02D-P61-R_11-22-06_Draft_SOS_Rpt-T1

Sacramento River Ecological Flows Study: State of the System Report

Public Review Draft

Prepared for
The Nature Conservancy
500 Main Street
Chico, CA 95928

by
Stillwater Sciences
2855 Telegraph Avenue, Suite 400
Berkeley, CA 94705

Funded by
CALFED Ecosystem Restoration Program

November 2006

Suggested Citation:
Stillwater Sciences. 2006. Sacramento River ecological flows study: State of the system report. Public Review draft. Prepared by Stillwater Sciences, Berkeley for The Nature Conservancy, California.

Executive Summary

This State of the System (SOS) Report is part of the Sacramento River Ecological Flows Study (the "Study") initiated by The Nature Conservancy (TNC) in collaboration with ESSA Technologies, Stillwater Sciences, UC Davis, and UC Berkeley. The study area focuses on the mainstem Sacramento River corridor between Keswick Dam (RM 302) and Colusa (RM 143), including the channel, adjacent floodplain and riparian habitats, and off-channel water bodies. The overarching goal of the Study is to define how flow characteristics (e.g., the magnitude, timing, duration, and frequency) and associated management actions (such as gravel augmentation and changes in bank armoring) influence the creation and maintenance of habitats for a number of native species that occur in the Sacramento River corridor. This SOS Report is designed to provide resource managers and stakeholders with information and tools that will allow them to explore how changes in the pattern of flow releases can affect habitats in the Sacramento River. In this way, the SOS Report should provide useful information for water operations planning, restoration planning, species recovery planning, and storage investigations (e.g., for Shasta Dam enlargement and the Sites Reservoir construction) that focus on the Sacramento River. The information in this SOS Report builds on the earlier review of Sacramento River ecological flow issues conducted by Kondolf et al. (2000) for CALFED.

The overall Sacramento River Ecological Flows Study is composed of four primary tasks. The first is this SOS Report, which is designed to distill existing information and present conceptual models and hypotheses about ecological flow needs in the Sacramento River. The second task is a series of field investigations and modeling applications designed to address data gaps and to refine estimates of ecological flow needs, including:

- a gravel study designed to characterize gravel quality, mobilization, and routing;
- an off-channel habitat study to estimate sediment deposition rates in, and resultant terrestrialization of, off-channel habitats; and
- a bank study to examine the effects of natural and rip-rapped banks on aquatic habitat;
- a numerical chute cutoff model to predict the flows required to create a chute cutoff;
- a refined meander migration model; and
- a sediment transport model that predicts the grain size distribution of both the surface and subsurface as a function of sediment supply and bed mobilization and scour.

The third task of the Study is a decision analysis tool, referred to as the Sacramento River Ecological Flows Tool (SacEFT), which is designed to facilitate the analysis of ecological trade-offs associated with different suites of management actions (ESSA Technologies 2005, 2006). The fourth and final task component will be a Final Report that summarizes and synthesizes the results of the field investigations, modeling applications, and the application and recommended future uses of the SacEFT.

The processes, habitats, and species of the Sacramento River have been the focus of much study, and the volume of available reports and datasets poses a challenge for synthesizing information and organizing a discussion of ecosystem components. Divergent conceptual models about process-habitat-biotic linkages complicate the process of summarizing what is known about the Sacramento River, and add to the challenge of evaluating alternative approaches for conserving and restoring the river ecosystem. To help overcome these challenges, this SOS Report discusses and analyzes the Sacramento River through the lens of six focal species. A focal species approach facilitates the exploration of linkages among ecosystem processes, resultant habitats, and biotic needs. For each focal species, we identify the different life history stages that occur in the Sacramento River, the habitats used by each of those life history stages, the ecological processes that create and maintain those habitats, and the management actions (e.g., changes in the flow regime or bank protection) that influence those ecological processes and habitat conditions. The six focal species selected for this SOS Report are:

- Chinook salmon (*Oncorhynchus tshawytscha),*
- steelhead (*Oncorhynchus mykiss*),
- green sturgeon (*Acipenser medirostros*),
- bank swallow (*Riparia riparia*),
- western pond turtle (*Clemmys marmorata*), and
- Fremont cottonwood (*Populus fremontii*).

This suite of focal species is not intended to be an exhaustive list of all conservation targets within the study area; rather, these six focal species provides a logical starting point for the Study while covering a wide range of habitats and ecological processes that occur in the Sacramento River. The loss and degradation of essential habitats in the Sacramento River corridor has generally reduced the river's capacity to support native species, assemblages, and guilds. Habitat impacts are discussed further in section 3 of this SOS Report. The six focal species discussed in this SOS Report help to highlight the effects of land use changes and water supply development on the broader ecosystem, and they highlight some of the key resource management challenges in the Sacramento River system.

Key findings and hypotheses of the SOS Report are summarized below, again using the focal species as a framework.

Chinook Salmon

While many rivers throughout the range of Chinook salmon (*Oncorhynchus tshawytscha)* support multiple runs, the Sacramento River is unique because it supports four distinct runs of Chinook salmon: winter-run, spring-run, fall-run, and late-fall-run.

Winter-run Chinook salmon

Though Chinook salmon range from California's Central Valley up north to Alaska, and west to the Kamchatka peninsula in Asia, the Sacramento River supports the only known population of winter-run Chinook salmon. Winter-run Chinook salmon are unique because they spawn during summer months when air temperatures usually approach their yearly maximum. As a result, winter-run Chinook require stream reaches with cold water sources that will protect embryos and juveniles from the warm ambient conditions in the summer. In addition to cold water temperature in the summer, winter-run Chinook salmon also require relatively warm water temperatures in the winter to promote fast growth of salmon fry to enhance survival and production. We hypothesize that this unique combination of cold summer water temperatures and warm winter water temperatures explains why winter-run Chinook salmon are found only in the Sacramento River.

We hypothesize that the life history strategy of winter-run Chinook salmon makes spawning habitat the most likely limiting factor for the population, both historically and currently, as discussed in section 4.2. Historically, the summer spawning and egg incubation stages restricted spawning to reaches that remain cold all summer, which were typically higher-elevation streams such as the McCloud River that were fed by cold water springs. These reaches are steeper and more confined than reaches downstream in the Sacramento Valley, so gravel resources were limited to small reaches and patches located within the predominately cobble and boulder bed. So, historically, winter-run Chinook were restricted not only in the linear extent of stream available to support spawning because of temperature constraints, but also in the amount of spawning gravel available to the population. Because winter-run Chinook spawn in late spring and early summer, their progeny emerge in late summer and early fall. No other salmonids in the Sacramento River emerge during this time, and most other juvenile salmonids outmigrate in the spring before summer water temperatures in the middle and lower Sacramento River become too warm. As a result, winter-run Chinook fry and juveniles had relatively little competition for rearing habitat in the fall and winter as they migrated downstream.

The construction and operation of Shasta Dam contributed to an initial increase in the winter-run salmon population by expanding the cumulative spawning habitat available to the population, as discussed in section 4.2. However, the positive effect of Shasta Dam on winter-run Chinook salmon began to wane in the late 1970s and early 1980s, when escapements reached dangerously low levels. The drought of 1976-77 caused a precipitous decline in winter-run escapements when lethally warm water temperatures were released into the Sacramento River. We hypothesize that, in addition to this precipitous decline, the progressive loss of spawning habitat caused by bed coarsening had already contributed to a decline in winter-run salmon escapements by the time of the drought, and that the loss of spawning habitat continued to limit the population through the 1980s, as discussed in section 4.2. There are several pieces of evidence to suggest that the loss of suitable pawning habitat has limited the winter-run salmon population. For example, two spawning habitat surveys conducted in 1964 and 1980 show a 54% loss of mapped spawning area between Keswick Dam (RM 302) and Anderson Bridge (RM 283). Similarly, the results of a new salmon population model developed by Stillwater Sciences provide a strong indication that spawning habitat is limiting winter-run salmon in the Sacramento River. We hypothesize that the recent increases in winter-run escapements can be attributed to improved fish passage at ACID Dam in 2001 and recent gravel augmentation implemented by the USBR between 1997 and 2001, both of which helped to increase spawning habitat for the population. Other factors have likely contributed to recent increases in escapements, such as reductions in ocean harvest and the increase in hatchery production of winter-run juveniles. However, the Stillwater Sciences salmon population model suggests that increases in spawning habitat have had a greater effect on the population increases.

The following suggestions emerge from this SOS Report and seek to maintain and enhance the recent increases in winter-run salmon escapements:

- Continue to augment the gravel supply in the upper Sacramento River, focusing injection between Keswick Dam (RM 302) and ACID Dam (RM 298.5) where winter-run spawning is currently concentrated;
- Consider operating the fish passage facilities at ACID Dam to force winter-run salmon to spawn downstream of the dam once the spawning habitat upstream of the dam is saturated;
- Expand gravel augmentation activities between ACID Dam (RM 298.5) and Clear Creek (RM 290) if the fish passage facilities at ACID Dam are used to re-distribute winter-run salmon spawning; and,
- Apply the USBR water temperature model to assess the potential effects of the proposed contraction of the cold water zone on winter-run fry rearing habitat and the potential distribution of predators in the upper Sacramento River.

Spring-run Chinook salmon

Though spring-run Chinook salmon were probably the most abundant salmonid in the Central Valley under historical conditions (Mills and Fisher 1994), large dams eliminated access to vast amounts of historical habitat and the spring run has suffered the most severe declines of any of the four Chinook runs in the Sacramento River basin (Fisher 1994). In the Sacramento River, the construction and operation of Shasta and Keswick dams also eliminated the spatial segregation of spring-run salmon spawning with that of fall-run salmon. Any attempt to restore a spawning population of spring-run salmon in the mainstem Sacramento River must restore the spatial segregation of spawning between spring-run and fall-run salmon; otherwise, the fall-run population would hybridize with and eclipse a restored spring-run population. Agencies have begun discussing creating a spring-run salmon "sanctuary" in the Sacramento River above ACID Dam (RM 298.5) by operating the dam's fish passage facilities to exclude fall-run salmon spawning.

The construction of Red Bluff Diversion Dam (RM 243) in 1966 also affected the mainstem spawning population of spring-run salmon. Spring-run Chinook display a stream-type life history strategy, because adults migrate upstream while sexually immature, hold in deep cold pools over the summer, and spawn in late summer and early fall. Migration studies conducted on winter-run Chinook salmon during the early 1980s found that upstream migrants often experienced migration delays at RBDD, ranging between 1 to 40 days, with an average delay of 18 days (Hallock and Fisher 1985). Current RBDD gate operations have the potential to block or impede the upstream migration for a portion of any spring-run population that is restored to the upper Sacramento River.

Though the mainstem spawning population of spring-run salmon has been virtually extirpated from the Sacramento River, there are currently three populations of spring-run salmon in Deer, Mill, and Butte Creeks. Restoration efforts are also being implemented to restore another spawning population of spring-run salmon in Battle Creek. In recent years, the Butte Creek population of spring-run salmon has increased significantly, but the Deer and Mill Creek populations have not exhibited a similar increase in escapements. We hypothesize that the increase in Butte Creek spring-run salmon escapements has been caused by seasonal inundation in the Sutter Bypass, which provides good quality fry rearing habitat that promotes faster fry growth and higher rates of fry survival. As a result of this seasonal inundation, the fry component contributes more to the spring-run population.

The following suggestions emerge from this SOS Report and seek to restore a spawning population of spring-run Chinook salmon to the mainstem Sacramento River, and to improve fry rearing habitat for both mainstem and tributary populations of spring-run:

- Consider operating the fish passage facilities at ACID Dam to create a spring-run salmon spawning sanctuary above the dam by excluding fall-run Chinook salmon;
- Continue to augment the gravel supply in the upper Sacramento River, focusing injection between Keswick Dam (RM 302) and ACID Dam (RM 298.5) where a restored population of spring-run salmon would be encouraged to spawn;
- Create a flood bypasses downstream of Deer and Mill creeks designed specifically to inundate annually to increase fry rearing habitat; and
- Increase the "gates out" period at RBDD until mid-June, or replace RBDD with an alternative water diversion structure.

Fall-run Chinook salmon

Fall-run Chinook salmon (*Oncorhynchus tshawytscha)* is the most abundant and widely distributed run of salmon in the Central Valley, in large measure because it has suffered relatively less displacement from historical habitats caused by dam construction. The relatively high abundance of fall-run Chinook salmon (or "fall Chinook") is also a function of hatchery supplementation, because fall Chinook have been the primary target of hatchery production at

Central Valley hatcheries for several decades. Despite the significantly higher abundance of fall Chinook relative to other salmonid populations, escapements have generally declined over the past few decades.

The success of the fall Chinook life history strategy is predicated on the production and survival of high numbers of juveniles, which requires abundant spawning habitat. The competition for spawning habitat is the most likely source of density-dependent mortality for fall-run Chinook salmon, as discussed in section 4.4. Because adult fall-run salmon spawn in the lower mainstem reaches of large rivers, where water temperatures may increase rapidly in the spring and summer, fall-run fry must emigrate quickly from fresh water at a relatively small size <3.5 in (90 mm) before water temperatures become stressful or lethal. Emigrating at such a small size makes fall-run fry vulnerable to predation; however, other components of the fall-run life history strategy helps to limit predation losses. For example, fall-run salmon fry and juveniles do not exhibit territorial behavior, which allows them to rear, smolt, and outmigrate in higher densities than other salmonids. By emigrating synchronously in schools, rather than as individuals, fall Chinook fry and smolts can swamp potential predators to avoid significant losses to predation, and by emigrating in late spring, they have the advantage of higher discharge fueled by early snowmelt, which can reduce their exposure to predation.

The following suggestions emerge from this SOS Report and seek to enhance escapements of fall-run Chinook salmon in the Sacramento River basin:

- Conduct a redd superimposition study to assess the degree to which it occurs during fall-run Chinook salmon spawning;
- Remove the coarse surface layer of armored beds between ACID Dam (RM 298.5) and Clear Creek (RM 290) to increase spawning habitat by exposing spawning-size gravel stored in the channel subsurface; and
- Augment the gravel supply of the upper Sacramento River between ACID Dam (RM 298.5) and Clear Creek (RM 290) to replenish the gravel scoured from the subsurface of the bed from patches where the coarse surface layer has been removed.

Late-fall-run salmon

The mainstem Sacramento River supports the largest spawning population of late-fall-run salmon, generally above RBDD (RM 243.5). Spawning populations of late-fall-run salmon also occur in several tributaries of the Sacramento River, including Battle, Cottonwood, Clear and Mill creeks, and the Feather and Yuba rivers (USFWS 1996). However, the sizes of these spawning populations are relatively small, with the exception of Battle Creek where late-fall-run Chinook are artificially propagated at the Coleman National Fish Hatchery (CNFH).

We hypothesize that the population of late-fall-run salmon in the Sacramento River is an artifact of Shasta Dam construction and operations. This is discussed further in section 4.5. Coldwater releases from Shasta Dam have changed the summer water temperature regime of the upper Sacramento River, effectively creating oversummering habitat where it did not exist before. As a result, late-emerging fall-run fry that historically would have perished from high water temperatures were able to survive by staying in the river to rear through the summer and then emigrating as yearlings the following fall. By supporting a yearling life history strategy, this oversummering habitat in the mainstem Sacramento River allowed the late-fall-run to emerge as a distinct run.

We hypothesize that the downstream limit of late-fall-run spawning is dictated by the summer water temperature regime. We also hypothesize that consistently low survival of the progeny of fish that spawn in reaches too far downstream exerted selective pressure over time, so that individuals of the population now spawn where summer water temperatures can support over-summering of juveniles.

The USBR recently proposed moving the water temperature compliance point to support winter-run salmon spawning and egg incubation. By moving the water temperature compliance point upstream to Ball's Ferry (RM 276), the USBR would have greater flexibility in managing Sacramento River water temperatures with the coldwater storage pool of Shasta Reservoir (USBR 2004). Our analyses suggest that the contraction of the cold water zone would likely have the greatest impact on the late-fall-run Chinook salmon population in the Sacramento River, by reducing the amount of oversummering habitat.

The following suggestions emerge from this SOS Report and seek to enhance escapements of late-fall-run Chinook salmon in the Sacramento River basin:

- Apply the USBR water temperature model to study the impacts of moving the water temperature compliance point on oversummering habitat for late-fall-run salmon juveniles;
- Remove the coarse surface layer of armored beds between ACID Dam (RM 298.5) and Clear Creek (RM 290) to increase spawning habitat by exposing spawning-size gravel stored in the channel subsurface; and
- Augment the gravel supply of the upper Sacramento River between ACID Dam (RM 298.5) and Clear Creek (RM 290) to replenish the gravel scoured from the subsurface of the bed from patches where the coarse surface layer has been removed.

Steelhead

The management of steelhead populations in Central Valley tributaries is usually subsumed within the management of Chinook salmon populations because of their similar life history strategies and habitat requirements. Nevertheless, steelhead generally exhibit a more flexible life history strategy than Chinook salmon, and the habitat requirements of juvenile steelhead differ from those of juvenile Chinook, as discussed in Section 5. The recreational harvest of rainbow trout in the upper Sacramento River indicates that rainbow trout and/or steelhead spawn in the mainstem Sacramento River, though preferred spawning locations are unclear. We hypothesize that rainbow trout and steelhead likely spawn above ACID Dam (RM 298.5); consequently, any proposal to operate the fish passage facilities of the dam to create a spring-run spawning sanctuary upstream has the potential to exclude steelhead from current spawning habitats.

Studies indicate that the majority of returning adult steelhead in the Central Valley spend two years in fresh water before emigrating to the ocean (McEwan 2001). For juvenile steelhead to survive the winter, they must avoid predation and high flows by finding cover and velocity refuge in the interstitial spaces between cobbles and boulders (Bjornn 1971, Hartman 1965, Bustard and Narver 1975, Swales et al. 1986, Everest et al. 1986, Grunbaum 1996). Age 0+ steelhead can use shallower habitats and can find interstitial cover in gravel-size substrates, while age 1+ or 2+ steelhead, because of their larger size, need coarser cobble/boulder substrate for cover (Bustard and Narver 1975; Bisson et al. 1982, 1988; Fontaine 1988; Dambacher 1991). As a result, we hypothesize that rearing habitat for age 1+ and 2+ juveniles is likely limiting populations of steelhead in the Sacramento River system.

The following suggestions emerge from this SOS Report and seek to improve understanding and to enhance the population of steelhead and rainbow trout that spawn in the mainstem channel of the upper Sacramento River:

- Conduct a survey of overwintering habitat for juvenile steelhead in the upper Sacramento River by sampling channel beds, armored banks and protected bridge pilings for particle size and interstitial space; and
- Place artificial structures in the upper Sacramento River to mimic overwintering habitat and monitor their use by juvenile steelhead during high flow events.

Green Sturgeon

Despite the importance of the Sacramento River as a spawning area for green sturgeon, little is known about specific spawning locations or habitat preferences. Current radio telemetry studies being conducted by UC Davis and private consultants may yield more detailed migratory and habitat data in the near future.

It appears that adult green sturgeon may spend up to six months in the Sacramento River. Adults loiter in the Bay-Delta estuary before beginning their upstream migration, which typically occurs between February and late July (Adams et al. 2002). The latter portion of the upstream migration period occurs when gates are closed at RBDD, so the dam may present a passage barrier for some green sturgeon spawners. In the Rogue River, upstream migrants hold in deep pools with slow velocities before migrating farther upstream to spawn, and then they hold in deep pools after spawning until the late fall or early winter, when they emigrate to downstream estuaries or the ocean (Erickson et al. 2002). Though it is not known if green sturgeon exhibit this pre- and post-spawning holding behavior in the Sacramento River, anecdotal evidence provided by anglers suggests that they do. During these holding phases, adult green sturgeon may be vulnerable to targeted or incidental harvest by anglers. Considering the apparently few number of spawners that migrate upstream to spawn each year, the river harvest of even a small number of adults may produce significant impacts on the population.

The following suggestions emerge from this SOS Report and seek to better understand and to enhance the population of green sturgeon that spawn in the mainstem Sacramento River:

- Increase the "gates out" period at RBDD until mid-June, or replace RBDD with an alternative water diversion structure, to facilitate upstream passage of adults;
- Consider season- and reach-specific angling restrictions to protect green sturgeon spawners holding in the vicinity of Hamilton City from targeted or incidental harvest;
- Supplement existing telemetry studies by capturing pre-spawning adults in the Sacramento River and fitting them with acoustic radio tags that can be tracked by hand and boat, to facilitate the identification of specific spawning habitats; and
- Apply the USBR water temperature model to evaluate the effects of moving the water temperature compliance point for winter-run Chinook salmon on green sturgeon larvae, using 59°F (15°C) as the water temperature target to be monitored.

Bank Swallow

There has been a general decline in the total number of bank swallow burrows, colonies, and estimated breeding pairs found between Redding and Verona (RM 292–81) since 1986. The Sacramento River and its tributaries provide important nesting locations for bank swallow, comprising approximately 70% of the bank swallow nesting in California (Hight 2000).

Bank swallow nests are built in burrows which the birds excavate in steeply sloped banks with friable soils (Garrison 1998, 1999). Most of California's hundred or so colonies occur in steep river banks and bluffs in the riparian ecosystems of large lowland rivers like the Sacramento River. Bank swallows generally benefit from bank erosion caused by high winter stream flow, which renews nesting habitat while they are in overwintering habitats to the south. Flood control and bank armoring projects directly threaten bank swallow nesting sites by destroying existing nesting habitat and by reducing or preventing bank erosion (B. Garrison and R. Schlorff 1997, Garrison 1998, Moffatt et al. 2005), as discussed in section 7.

High flows during nesting season are generally infrequent on the Sacramento River but nevertheless have the potential to adversely affect bank swallow colonies. Although there is general disagreement on the exact magnitude of flow required to initiate substantial bank erosion, there is growing evidence that flows in the 20,000–25,000 cfs range can begin to erode some banks, causing partial bank collapse that can result in localized nest failure if swallows are present. Flows above 50,000–60,000 cfs are almost certain to cause widespread bank erosion, leading to partial or complete colony failure at many sites if breeding bank swallows are present.

The installation of riprap and concrete in bank armoring activities can have the immediate effect of reducing the availability of sufficiently steep, suitably textured habitat for bank swallow nesting colonies. Overall, an estimated 48% of the channel from Red Bluff to Colusa (RM 243–143) is now covered by riprap on at least one side (Larsen and Greco 2002, S. Greco, unpublished data). However, bank protection has been preferentially applied to actively migrating bends which would otherwise be among the most suitable sites for bank swallow nests. Hence, it is likely that bank protection has eliminated substantially more than 48% of potential nesting sites between Red Bluff and Colusa. Plans for new bank protection projects on the Sacramento River continue to be developed. If implemented, these projects would further reduce available habitat, with an extremely detrimental cumulative effect on the bank swallow population (Schlorff 2004).

A levee removal and rip-rap retirement project was completed on the mainstem Sacramento River at RM 233 in late fall 1999 (Golet et al. 2003). Erosion induced by winter storms expanded an existing cut bank, and a swallow colony was established there in the spring of 2000. The newly established colony, with 2,770 burrows, was the largest on the river that year. It represented a substantial expansion for bank swallows at the site, which had supported just 930 burrows in the previous year.

The following suggestions emerge from this SOS Report and seek to better understand and to enhance the populations of bank swallow that nest in the mainstem Sacramento River corridor:

- Manage high flow releases to promote natural patterns of bank erosion in the middle Sacramento River before the beginning of the breeding season (late March) and minimize flow events with magnitudes over 20,000 cfs beginning in late March to prevent bank sloughing and nest inundation during the nesting season;
- Conduct a GIS analysis to identify potential meander migration into soils of suitable texture that would support bank swallow nesting, and use the analysis to prioritize conservation or bank armor removal projects;
- Identify and implement opportunities to retire bank armoring in locations where meander migration is projected to occur in appropriately textured soils to support nesting colonies;
- Identify locations where levee setbacks can create a larger meander zone to promote natural bank erosion processes and increase potential nesting habitat.

Western Pond Turtle

The western pond turtle (*Clemmys marmorata*) is California's only native freshwater turtle. Western pond turtles rely on habitat types (e.g., oxbow lakes) that have relatively slow rates of formation. The creation of new off-channel water bodies generally requires several high flow events that drive the processes of meander migration and channel cutoff, but these high flow events happen only periodically. Similarly, off-channel water bodies gradually terrestrialize as they fill with sediment and organic detritus, and as they are colonized by vegetation. Consequently, activities that promote the formation of off-channel water bodies (e.g., levee setback, retirement of bank armor, retaining aspects of the natural flow regime) are key to maintaining this important type of habitat.

Western pond turtle breeding activity peaks in May through July, but may occur throughout the year (Holland 1994, Reese 1996). The incubation period for western pond turtle eggs averages 80 days (mainly starting in June–July), but in some cases may exceed 100 days in California (Bettelheim 2005). Incubating eggs are extremely sensitive to increased soil moisture, which can cause high mortality (Bettelheim 2005, Shaffer 2005, Ashton et al. 1997). In wet conditions, eggs can literally explode from internal pressure caused by water absorption (Ashton et al. 1997). Consequently, the timing of natural inundation, irrigation, and regulated high flows during the summer incubation period is an important factor to track in order to assess the effects of management and restoration activities on western pond turtle.

The following suggestions emerge from this SOS Report and seek to better understand and protect western pond turtles in the Sacramento River corridor:

- Conduct basic surveys to assess the abundance and distribution of western pond turtle;
- Manage flow regime patterns to promote the natural processes of bank erosion, meander migration, and channel cutoff in the middle Sacramento River;
- Identify and implement opportunities to retire bank armoring and/or setback levees in the middle Sacramento River; and
- Manage flows during the primary summer nesting season to reduce risk of nest inundation by minimizing flow fluctuations that cause a substantial increase in river stage (and associated inundation of floodplain sites that might contain turtle nests).

Fremont Cottonwood

In the Central Valley, Fremont cottonwood (*Populus fremontii* ssp. *fremontii*) is the tree species that dominates the near-river forests. Soon after establishment, it provides ecological structure to the riparian ecosystem by stabilizing the substrate, fixing carbon, generating large woody debris, and creating vertical stratification for wildlife habitat.

The Sacramento Valley has lost about 98.5% of riparian forests since 1850 (Katibah 1984, Greco 1999). The greatest extent of cottonwood forest remaining along the Sacramento River occurs between Red Bluff (RM 245) and Colusa (RM 143) (Vaghti and Greco, in press, SRCAF 2003).

Recent work at three point bars between RM 192-172 indicates that successful cottonwood recruitment occurs at relative elevations of 3–9 ft (1–3 m) above summer baseflow levels (Roberts et al. 2002, TNC 2003a). Similar results have been observed along the lower Tuolumne and San Joaquin rivers, although successful recruitment band in these smaller rivers tends to be at slightly lower elevations of 2–6 ft (0.6–2 m) (McBain and Trush 2002, Stillwater Sciences 2003 and 2006, Stella 2005).

Because willows and cottonwood seedlings are phreatophytic (i.e., their roots must maintain contact with a perennial water source), they are vulnerable to desiccation at steep rates of water table decline. Recent field studies on Sacramento River point bars (TNC 2003, Morgan 2005, Morgan and Henderson 2005a and b) indicate that successful establishment of large cohorts of Fremont cottonwood seedlings is most likely to occur when water table/river stage declines at average rates of less than 0.8 in/day (2 cm/day) (Stella et al. in review, Stillwater Sciences 2006, Stella 2005, Morgan 2005, Morgan and Henderson 2005b). These same studies indicate that rates of decline in the range of 0.8 to 1.6 in/day (2-4 cm/day) are stressful to seedlings, but may still support survival of a smaller cohort of seedlings. It is also possible that steeper rates of river stage recession may be acceptable if they are offset by periods of 1 or more days of stable water levels, which would produce a stepped recession limb of the recruitment flow hydrograph (TNC 2003a, Stillwater Sciences 2006).

Reductions in the magnitude and frequency of winter overbank flows in the post-dam era have presumably led to an overall decrease in soil moisture during the growing season for cottonwoods and other riparian plants. We hypothesize that this has contributed to reduced growth rates and altered competitive interactions such that species more tolerant of somewhat drier conditions may become more dominant. This may have led to an increase in the abundance of box elder and walnut (see Wood 2003b, Vaghti 2003, Fremier 2003).

The reduced magnitude, and possibly altered timing, of spring flows may have also affected cottonwoods by encouraging recruitment on low depositional surfaces that become inundated by subsequent winter floods or by elevated summer baseflows (Morgan 2005, Morgan and Henderson 2005ab), as discussed in section 9.

Morgan (2005) concluded that there were three primary attributes of the current altered hydrograph that limit cottonwood seedling survival at the middle Sacramento River study sites: 1) the reversal of summer flows such that there is now an increasing trend in summer flow levels during cottonwood seed release and seed germination periods, 2) rapid stage declines during the spring pulse flow such that root growth in seedlings established during the typical recruitment period cannot keep up with declining water levels, and 3) the immediate drop in stage late in growing season when reservoir releases for summer irrigation cease.

To promote riparian vegetation recruitment and establishment in the Sacramento River corridor, we recommend that agencies:

- Manage the recession limb of spring high flow events in wet water years to promote seedling establishment of cottonwoods and willows;
- Promote channel migration to create new seedbeds for cottonwood recruitment through scour and fine sediment deposition;
- Promote strategic horticultural restoration on higher floodplains surfaces where passive recruitment is infeasible; and
- Prioritize actions to eradicate and control invasive plant species.

Synthesis

A number of field studies, modeling efforts, and decision analyses will further inform the initial recommendations provided here. This additional work will be summarized in a Final Report at the end of the Study (projected for summer 2007).

The Sacramento River Ecological Flows Study was initiated to evaluate restoration strategies that might complement revegetation activities implemented by several groups along the river. The view of stream flow as the "master" variable regulating form and function of riverine habitats is shared by a growing body of researchers, both in California and worldwide, who are focusing on understanding how riverine ecosystems are affected by changes in parameters such as the frequency, magnitude, timing, duration, and rate of change of flow. By improving the understanding of linkages between flow characteristics and ecological goals, the Sacramento River Ecological Flows Study seeks to provide information to support future decision making and water planning.

For the current recommendations included here, we have made an effort to develop recommendations that are both effective and feasible. Some could be implemented with relative ease while others would entail a more involved process and possible further study. The actions are also aligned with the approach of restoring the processes that create and maintain habitats on the Sacramento River.

Opportunities to integrate the information provided in this SOS Report are numerous and include the proposed Sites Reservoir off-stream storage facility and the proposed raising of Shasta Dam, among others. Understanding the operational impacts and potential opportunities of each of these projects will require improved understanding of the Sacramento River ecosystem, and could help inform a multiple-benefit project design that, under ideal conditions, would meet human water demands while providing ecological benefits.

It is our goal that resource managers working on the Sacramento River will benefit from the information provided in this SOS Report and that all interested in the Sacramento River will be able to strike a balance between ecosystem and human needs dependent on the river.

Executive Summary References

Adams, P. B., C. B. Grimes, J. E. Hightower, S. T. Lindley, and M. L. Moser. 2002. Status review for North American green sturgeon, *Acipenser medirostris.* National Marine Fisheries Service, Santa Cruz, California.

Ashton, D. T., A. J. Lind, and K. E. Schlick. 1997. Western pond turtle (*Clemmys marmorata*). Natural history. USDA Forest Service, Pacific Southwest Research Station, Redwood Sciences Laboratory, Arcata, California.

Bettelheim, M. P. 2005. The western pond turtle, *Clemmys marmorata*: a natural history of the species. Walnut Creek, California.

Bisson, P. A., K. Sullivan, and J. L. Nielsen. 1988. Channel hydraulics, habitat use, and body form of juvenile coho salmon, steelhead trout, and cutthroat trout in streams. Transactions of the American Fisheries Society 117: 262-273.

Bisson, P., J. L. Nielsen, R. A. Palmason, and L. E. Grove. 1982. A system of naming habitat types in small streams, with examples of habitat utilization by salmonids during low streamflows. Pages 62-73 in N. B. Armantrout, editor. Proceedings of the symposium on acquisition and utilization of aquatic habitat inventory information. American Fisheries Society, Western Division, Bethesda, Maryland.

Bjornn, T. C. 1971. Trout and salmon movements in two Idaho streams as related to temperature, food, stream flow, cover, and population density. Transactions of the American Fisheries Society 100: 423-438.

Bustard, D. R., and D. W. Narver. 1975. Aspects of the winter ecology of juvenile coho salmon (*Oncorhynchus kisutch*) and steelhead trout (*Salmo gairdneri*). Journal of the Fisheries Research Board of Canada 32: 667-680.

Dambacher, J. M. 1991. Distribution, abundance, and emigration of juvenile steelhead (*Oncorhynchus mykiss*), and analysis of stream habitat in the Steamboat Creek basin, Oregon. Master's thesis. Oregon State University, Corvallis.

Erickson, D. L., J. A. North, J. E. Hightower, J. Weber, and L. Lauck. 2002. Movement and habitat use of green sturgeon *Acipenser medirostris* in the Rogue River, Oregon, USA. Journal of Applied Ichthyology 18: 565-569.

ESSA Technologies Ltd. 2005. Sacramento River Decision Analysis Tool: workshop backgrounder. Prepared by ESSA Technologies Ltd., Vancouver, British Columbia for The Nature Conservancy, Chico, California.

ESSA Technologies Ltd. 2006. Sacramento River Ecological Flows Tool (SacEFT): design guidelines. Prepared for by ESSA Technologies Ltd., Vancouver, British Columbia for The Nature Conservancy, Chico, California.

Everest, F. H., G. H. Reeves, J. R. Sedell, J. Wolfe, D. Hohler, and D. A. Heller. 1986. Abundance, behavior, and habitat utilization by coho salmon and steelhead trout in Fish Creek, Oregon, as influenced by habitat enhancement. Annual Report 1985 Project No. 84-11. Prepared by U. S. Forest Service for Bonneville Power Administration, Portland, Oregon.

Fisher, F. W. 1994. Past and present status of Central Valley Chinook salmon. Conservation Biology 8: 870-873.

Fontaine, B. L. 1988. An evaluation of the effectiveness of instream structures for steelhead trout rearing habitat in the Steamboat Creek basin. Master's thesis. Oregon State University, Corvallis.

Fremier, A. K. 2003. Floodplain age modeling techniques to analyze channel migration and vegetation patch dynamics on the Sacramento River, California. Master's thesis. University of California, Davis.

Garrison, B. A. 1998. Bank swallow (*Riparia riparia*). *In* California Partners in Flight Riparian Bird Conservation Plan: a strategy for reversing the decline of riparian-associated birds in California. California Partners in Flight, Point Reyes Bird Observatory, Stinson Beach, California. http://www.prbo.org/calpif/htmldocs/species/riparian/bank_swallow_acct2.html

Garrison, B. A. 1999. Bank swallow (*Riparia riparia*). No. 414. *In* A. Poole and F. Gill, editors. The birds of North America. The Academy of Natural Sciences, Philadelphia, Pennsylvania and The American Ornithologists' Union, Washington, D. C.

Golet, G. H., D. L. Brown, E. E. Crone, G. R. Geupel, S. E. Greco, K. D. Holl, D. E. Jukkola, G. M. Kondolf, E. W. Larsen, F. K. Ligon, R. A. Luster, M. P. Marchetti, N. Nur, B. K. Orr, D. R. Peterson, M. E. Power, W. E. Rainey, M. D. Roberts, J. G. Silveira, S. L. Small, J. C. Vick, D. S. Wilson, and D. M. Wood. 2003. Using science to evaluate restoration efforts and ecosystem health on the Sacramento River Project, California. Pages 368-385 *in* P. M. Faber, editor. California riparian systems: processes and floodplain management, ecology, and restoration. 2001. Riparian habitat and floodplain conference proceedings. Riparian Habitat Joint Venture, Sacramento, California.

Greco, S. E. 1999. Monitoring riparian landscape change and modeling habitat dynamics of the yellow-billed cuckoo on the Sacramento River, California. Doctoral dissertation. University of California, Davis.

Grunbaum, J. B. 1996. Geographical and seasonal variation in diel habitat use by juvenile (age 1+) steelhead trout (*Oncorhynchus mykiss*) in Oregon coastal and inland streams. Master's thesis. Oregon State University, Corvallis.

Hallock, R. J., and F.W. Fisher. 1985. Status of winter-run Chinook salmon, *Oncorhynchus tshawytscha*, in the Sacramento River, California. AFB Office Report, January 25, 1985. California Department of Fish and Game, Anadromous Fisheries Branch.

Hartman, G. F. 1965. The role of behavior in the ecology and interaction of underyearling coho salmon (*Oncorhynchus kisutch*) and steelhead trout (*Salmo gairdneri*). Journal of the Fisheries Research Board of Canada 22: 1035-1081.

Hight, R. C. 2000. Annual report on the status of the threatened bank swallow. California Department of Fish and Game.

Holland, D. C. 1994. The western pond turtle: habitat and history. Final Report DOE/BP-62137-1. Bonneville Power Administration, Portland, Oregon.

Katibah, E. F. 1984. A brief history of riparian forests in the Central Valley of California. Pages 23-29 *in* R. E. Warner and K. M. Hendrix, editors. California riparian systems: ecology, conservation, and productive management. University of California Press, Berkeley.

Kondolf, G. M., T. Griggs, E. W. Larsen, S. McBain, M. Tompkins, J. G. Williams, and J. Vick. 2000. Flow regime requirements for habitat restoration along the Sacramento River between Colusa and Red Bluff. CALFED Bay Delta Program, Integrated Storage Investigation, Sacramento, California.

Larsen, E. W., and S. E. Greco. 2002. Modeling channel management impacts on river migration: a case study of Woodson Bridge State Recreation Area, Sacramento River, California, USA. Environmental Management 30: 209-224.

McBain and Trush. 2002. San Joaquin River restoration study background report. Prepared for Friant Water Users Authority, Lindsay, California and Natural Resources Defense Council, San Francisco, California by McBain and Trush, Arcata.

McEwan, D. 2001. Central Valley Steelhead. In Contributions to the Biology of Central Valley Salmonids, Fish Bulletin 179. California Dept. of Fish and Game.

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

Moffatt, K. C., E. E. Crone, K. D. Holl, R. W. Schlorff, and B. A. Garrison. 2005. Importance of hydrologic and landscape heterogeneity for restoring bank swallow (*Riparia riparia*) colonies along the Sacramento River, California. Restoration Ecology 13: 391-402.

Morgan, T. L. 2005. Hydrological and physiological factors controlling Fremont cottonwood seedling establishment along the Sacramento River, California: surface and alluvial groundwater relations, ecophysiological analysis from 2002-2004 field seasons, and cottonwood water use determination using stable isotope analysis. Master's thesis. California State University, Chico.

Morgan, T. L., and A. Henderson. 2005a. Field observations of cottonwood seedling survival at River Mile 192.5 during 2002 and 2003, Sacramento River, California. 30 March. Memorandum report. Prepared for California Department of Water Resources, Northern District.

Morgan, T. L., and A. Henderson. 2005b. Cottonwood seedling monitoring during 2004 and 2005 along the Sacramento River, California. 30 December. Memorandum draft report. Prepared for California Department of Water Resources, Northern District.

Nature Conservancy, The. 2003. Beehive Bend Subreach addendum to: A pilot investigation of cottonwood recruitment on the Sacramento River. The Nature Conservancy, North Central Valley Office, Chico, California.

Reese, D. A. 1996. Comparative demography and habitat use of western pond turtle in northern California: the effects of damming and related alterations. Doctoral dissertation. University of California, Berkeley.

Roberts, M. D., D. R. Peterson, D. E. Jukkola, and V. L. Snowden. 2002. A pilot investigation of cottonwood recruitment on the Sacramento River. The Nature Conservancy, Sacramento River Project, Chico, California.

Schlorff, R. W. 1997. Monitoring bank swallow populations on the Sacramento River: a decade of decline. Transactions of the Western Section of the Wildlife Society 33: 40-48.

Schlorff, R. W. 2004. Bank swallow population surveys, Sacramento River. Internal memoranda for files. California Department of Fish and Game, Habitat Conservation Division, Resource Assessment Division, Sacramento.

Shaffer, H. B. 2005. Survival of pond turtles in modified waterways: how can it work, and why does it matter? Western pond turtle workshop: ecology and conservation. The Wildlife Society, San Francisco Bay Area Chapter.

SRCAF (Sacramento River Conservation Area Forum). 2003. Sacramento River Conservation Area Forum handbook. Prepared for The Resources Agency, State of California by the Sacramento River Advisory Council; revised and updated by the Sacramento River Conservation Area Forum, Red Bluff, California. http://www.sacramentoriver.ca.gov.

Stillwater Sciences. 2003. Draft restoration strategies for the San Joaquin River: adaptive management report. Prepared for Natural Resources Defense Council, San Francisco, California and Friant Water Users Authority, Lindsay, California by Stillwater Sciences, Berkeley, California.

Stillwater Sciences. 2006. Restoring recruitment processes for riparian cottonwoods and willows: a field-calibrated predictive model for the lower San Joaquin Basin. Prepared for CALFED Bay-Delta Ecosystem Restoration Program, Sacramento, California by J. Stella, Stillwater Sciences, Berkeley, California, in conjunction with J. Battles and J. McBride, Department of Environmental Science, Policy, and Management, University of California, Berkeley.

Stella, J. C. 2005. A field-calibrated model of pioneer riparian tree recruitment for the San Joaquin Basin, CA. University of California, Berkeley, Berkeley, CA.

Stella, J. C., J. J. Battles, J. R. McBride, B. K. Orr. In review. Survival, growth, and physiology of riparian tree seedlings in response to simulated river stage decline. Submitted to Tree Physiology.

Swales, S., R. B. Lauzier, and C. D. Levings. 1986. Winter habitat preferences of juvenile salmonids in two interior rivers in British Columbia. Canadian Journal of Zoology 64: 1506-1514.

USBR (U. S. Bureau of Reclamation). 2004. Long-term Central Valley Project and State Water Project Operations Criteria and Plan Biological Assessment. USDI Bureau of Reclamation, Mid-Pacific Region, Sacramento, California.

USFWS (U. S. Fish and Wildlife Service). 1996. Recovery plan for the Sacramento-San Joaquin Delta native fishes. U. S. Fish and Wildlife Service, Region 1, Portland, Oregon.

Vaghti, M. G. 2003. Riparian vegetation classification in relation to environmental gradients, Sacramento River, California. Master's Thesis, University of California, Davis.

Vaghti, M. G., and S. E. Greco. In press. Riparian vegetation of the Great Valley. *In* Terrestrial Vegetation of California. Third edition.

Wood, D. M. 2003. The distribution and composition of woody species in riparian forests along the middle Sacramento River, California. Research and Long-term Monitoring Implementation: Beehive Bend and Chico Landing Sub-reaches. Prepared by Department of Biological Sciences, California State University, Chico for The Nature Conservancy, Sacramento River Project, Chico, California.

Preface

This State of the System (SOS) Report is part of the Sacramento River Ecological Flows Study (the "Study"), which was initiated by The Nature Conservancy (TNC) in collaboration with ESSA Technologies, Stillwater Sciences, U.C. Davis, and U.C Berkeley. To provide context for the report, we provide a brief overview of the overall project below.

Prior to Indo-European colonization, approximately 500,000 ac (200,000 ha) of riparian and upland forest flanked the Sacramento River in swaths as wide as 5 mi (8 km); however, this habitat has been reduced by nearly 95% over the past 150 years. TNC's Sacramento River Project team and its partners have worked for nearly two decades to restore natural ecosystem function within extensive tracts of the riparian corridor of the Sacramento River, one of California's most important rivers. Restoration strategies to date have focused on active revegetation of the floodplain to provide an immediate local increase in ecological value and thus ameliorate the problem of habitat fragmentation. Results of several studies confirm that it is possible to rapidly improve ecological conditions via this strategy as channel and floodplain habitats in restored reaches are utilized by endangered and threatened species.

Organizations and agencies involved in the conservation of the Sacramento River have concentrated their efforts between Red Bluff and Colusa, where natural ecological processes such as lateral river migration continue to operate to some degree. The natural dynamics of this reach make it likely to respond desirably to the restoration projects. The choice of the Red Bluff to Colusa reach reflects recognition of the fact that long-term conservation of key Sacramento River habitats will need to focus on restoring or replicating the natural processes that create and maintain dynamic riverine ecosystems.

The Sacramento River Ecological Flows Study was initiated to evaluate restoration strategies that might complement the revegetation activities of previous and ongoing efforts. The view of stream flow as the "master" variable regulating form and function of riverine habitats is shared by a growing body of researchers, both in California and worldwide, who are focusing on understanding how riverine ecosystems are affected by changes in parameters such as the frequency, magnitude, timing, duration, and rate of change of flow. By improving the understanding of linkages between flow characteristics and ecological goals, the ecological flows project seeks to help integrate more informed decision making into future water management strategies.

Opportunities to integrate this type of information are numerous and include the proposed Sites Reservoir off-stream storage facility, the proposed raising of Shasta Dam, new diversion and water transfer projects, and the Bureau of Reclamation re-consultation for the Operations Criteria and Plan (OCAP) for management of the Central Valley Project (which has resulted from the recent listing of green sturgeon and analysis of critical habitat for steelhead and spring-run Chinook salmon). Similarly, projects in the Sacramento – San Joaquin Bay-Delta that could alter flows on the Sacramento River are also under consideration. Understanding the operational impacts and potential opportunities of each of these projects will require improved understanding of the Sacramento River ecosystem, and could help inform a multiple benefit project design that, under ideal conditions, would meet human water demands while providing ecological benefits.

To help fill the ecological data gap and move the multiple benefits concept forward, TNC formed a team of ecologists and river management specialists and submitted a proposal to the CALFED Ecosystem Restoration Program (CALFED) in 2001. After extensive reviews by CALFED, independent technical reviewers, and individual stakeholders, the proposal was accepted and funded by CALFED in 2004 (Grant # ERP-02D-P61).

The Study seeks to advance understanding of how river flow affects key ecological indicators, including rates and patterns of natural processes and the abundance and distribution of native species and natural vegetation. The objective is not to return the system to a completely natural flow regime but rather to determine which elements of the natural conditions must be in place to promote a healthy ecosystem while meeting human needs. The Study has the following goals:

1. Synthesize existing interdisciplinary information on linkages among habitats, biota, and hydro-geomorphic processes,
2. Develop a decision analysis tool to evaluate trade-offs among different ecological objectives.
3. Propose strategies to achieve multiple species conservation benefits.
4. Provide information on ecological flow needs to other efforts seeking to balance ecosystem and human needs related to river flow.

To meet these goals, the Study team developed the following tasks and products:

1. ***Synthesize existing information***

   There is a great amount of information about the Sacramento River, but different reports focus on different species, different reaches of river, and different issues. The report presented here includes a summary and synthesis of a large amount of this information. The synthesis provided by this SOS Report is designed to stimulate new hypotheses. An important secondary function of the SOS report is to inform the development of models that may be used in constructing the SacEFT (see below).

2. ***Conduct a technical workshop to develop a preliminary Sacramento River Ecological Flows Tool (SacEFT)***

   The SacEFT will contain an integrated computer database for evaluation of ecological trade-offs of a variety of management actions. Ideally, it will serve as an ecological “plug-in” to existing water management models such as CALSIM II.

3. ***Conduct a stakeholder workshop to review and contribute to a report describing the "State of the Sacramento River System"***

   This SOS Report has benefited from the help of numerous experts who have contributed reports, ideas, and feedback in small meetings, telephone conversations, and email exchanges. To facilitate additional feedback and to widen the circle of expert contributors, the Study team will conduct a public workshop to solicit comment on a draft of this SOS Report.

4. ***Conduct a series of five field and computer simulation studies to address uncertainties and produce study plans and technical memos summarizing the findings***

   The study plans will guide targeted field studies including: investigations of salmon spawning and rearing habitat, gravel quality and thresholds of movement, meander migration, and the formation, evolution, and biological value of oxbow slough habitats. Most of the Study Plans were formulated in response to needs identified in a previous CALFED Integrated Storage Investigation report (Kondolf et al. 2000). Technical memos will summarize the findings of these studies, explain their relevance to resource management, and eventually be integrated into a final report.

5. ***Develop a new sediment transport model, finalize the SacEFT, and use these new tools to evaluate flow-related strategies in partnership with CALFED***.

   The new sediment transport model was developed to investigate differences in size distributions of surface and sub-surface material in coarse sediment deposits. Quantifying these differences is key to understanding the usability of sediment for spawning salmon. The SacEFT will be used to evaluate collaboratively developed management scenarios involving changes in the flow regime. Other actions such as altering bank armoring, levee alignments, or gravel augmentations may also be evaluated with this tool.

6. ***Complete reporting and conduct outreach; conduct a mid-project and final stakeholder review workshop; and release a final report***

   The final report will integrate the findings of the field and modeling studies, explain how new data assisted in the development of the management scenarios, and present the results and conclusions of the SacEFT evaluations.

This SOS Report characterizes the state of the mainstem Sacramento River with respect to a number of conservation targets. Ideally, this SOS Report will help inform water planning forums and make it easier to include ecological benefits among the operation goals of future water development projects.

Any inquiries or comments regarding this report should be directed to:

Ryan Luster
The Nature Conservancy
500 Main St., Chico, CA 95928
(530) 897-6370, ext. 213
rluster@tnc.org

Table of Contents

List of Tables

List of Figures

Definitions of Abbreviations, Acronyms, Initialisms

Term	Definition
ACID	Anderson Cottonwood Irrigation District
AFRP	Anadromous Fish Restoration Project
BLM	U.S. Department of Interior, Bureau of Land Management
CCVS	California Central Valley Steelhead
CDFG	California Department of Fish and Game
CDWR	California Department of Water Resources
CESA	California Endangered Species Act
CSU	California State University
CVP	Central Valley Project
CVPIA	Central Valley Project Improvement Act
D_{50}	Median particle size diameter
D_{84}	Particle size diameter that equals or exceeds 84 percent of the streambed particles
D_g	Geometric mean particle size diameter
DPS	Distinct Population Segment
EIS/EIR	Environmental Impact Report/Environmental Impact Statement
ESA	Federal Endangered Species Act
ESU	Evolutionarily Significant Unit
GCID	Glenn-Colusa Irrigation District
GIS	Geographic Information System
HSI	Habitat suitability index
LWD	Large woody debris
NA	Not applicable
NMFS	National Marine Fisheries Service
NOAA	National Oceanic and Atmospheric Administration
NODOS	North-of-the-Delta-Offstream-Storage
OCAP	Operations Criteria and Plan
PED	Pre-construction engineering and design
Q	Discharge
RBDD	Red Bluff Diversion Dam
RST	Rotary screw trap
SacEFT	Sacramento River Ecological Flows Tool
SI	Suitability index
SOS	State of the System
SRA	Shaded riverine aquatic
SWP	State Water Project
TAG	Technical Advisory Group
TCD	Temperature Control Devise
TNC	The Nature Conservancy
TRT	Technical Recovery Team
TUGS	The Unified Gravel and Sand model
UCD	University of California, Davis
USACE	U.S. Department of Defense, Army Corps of Engineers
USBR	U.S. Department of Interior, Bureau of Reclamation
USFWS	U.S. Department of Interior, Fish and Wildlife Service
USGS	U.S. Department of Interior, Geological Survey

Definitions of Units and Measurements

Unit	Definition
%	Percent
ac	Acre
af	Acre-foot
°C	Degrees Celsius or centigrade
cfs	Cubic feet per second
cm	Centimeter
°F	Degrees Fahrenheit
ft	Foot or feet
ha	Hectare
in	Inch
km	Kilometer
km^2	Square kilometers
km^3	Cubic kilometers
m	Meter
m^3	Cubic meter
mi	Mile
mi^2	Square miles
mm	Millimeter
RM	River mile
sec	Second
yd^3	Cubic yard
yr^{-1}	Per year

Conversion Factors

The table below is provided to allow ease of conversion for most units of measure used in this report.

Quantity	**English unit (abbreviation)**	**Metric unit (abbreviation)**	**To convert metric unit to English unit multiply metric unit by:**	**To convert English unit to metric unit multiply English unit by:**
Length	inches (in)	millimeters (mm)	0.03937	25.4
	inches (in)	centimeters (cm)	0.3937	2.54
	feet (ft)	meters (m)	3.2808	0.3048
	yards (yd)	meters (m)	1.094	0.9144
	miles (mi)	kilometers (km)	0.62139	1.6093
Area	square feet (ft^2)	square meters (m^2)	10.764	0.092903
	square miles (mi^2)	square kilometers (km^2)	0.3861	2.59
	acres (ac)	Hectares (ha)	2.47105	0.40469
Volume	cubic feet (ft^3)	cubic meters (m^3)	35.315	0.028317
	cubic yards (yd^3)	cubic meters (m^3)	1.308	0.76455
	acre-feet (ac-ft)	cubic decameters (dam^3)	0.8107	1.2335
Flow	cubic feet per second (cfs)	not converted	NA	NA
Velocity	feet per second (ft/s)	meters per second (m/s)	3.2808	0.3048
Temperature	degrees Fahrenheit (°F)	degrees Celsius (°C)	(1.8x°C)+32	(°F-32)/1.8

1 INTRODUCTION AND PURPOSE

1.1 Report Purpose

This State of the System (SOS) Report is one part of the Sacramento River Ecological Flows Study (the "Study") initiated by The Nature Conservancy (TNC), in collaboration with ESSA Technologies, Stillwater Sciences, UC Davis, and UC Berkeley. This SOS Report is designed to fulfill several objectives, including:

- to distill existing information about fundamental ecosystem processes, habitats, and native species in the mainstem Sacramento River;
- to refine conceptual models that illustrate key linkages between physical processes, habitat conditions, and biological responses of key native species; and
- to examine how flow characteristics (e.g., the magnitude, timing, duration, and frequency) and associated management actions (e.g., gravel augmentation and changes in bank armoring) influence the creation and maintenance of habitats for several native species that occur in the Sacramento River corridor.

This SOS Report is designed to provide resource managers and stakeholders with information and tools that will allow them to explore how changes in the pattern of flow releases can affect habitats in the Sacramento River. In this way, the SOS Report should provide useful information for water operations planning, restoration planning, species recovery planning, and storage investigations (e.g., for Shasta Dam enlargement and the Sites Reservoir construction) that are focusing upon the Sacramento River.

1.2 Study Tasks and Background

The overall Study is composed of four primary tasks. The first is this SOS Report, which is designed to distill existing information and present conceptual models and hypotheses about ecological flow needs in the Sacramento River. The second task is a series of field investigations and modeling applications designed to address data gaps and refine estimates of ecological flow needs. The information in this SOS Report builds on the earlier review of Sacramento River ecological flow issues conducted by Kondolf et al. (2000) for CALFED. In addition, some of the field studies currently being conducted for the Sacramento River Ecological Flows Study were designed to address some of the key uncertainties identified in the Kondolf et al. (2000) report.

The field studies include a gravel study designed to characterize gravel quality, mobilization, and routing; an off-channel habitat study to estimate sediment deposition rates in, and resultant terrestrialization of, off-channel habitats; and a bank study to examine the effects of natural and rip-rapped banks on aquatic habitat. Modeling applications include the development of a numerical chute cutoff model to predict the flows required to create a chute cutoff; the refinement and application of a meander migration model; and the development of a sediment transport model that predicts the grain size distribution of both the surface and subsurface as a function of sediment supply and bed mobilization and scour.

The third task of the Study is a decision analysis tool, referred to as the Sacramento River Ecological Flows Tool (SacEFT), which is designed to facilitate the analysis of ecological trade-offs associated with different suites of management actions (ESSA Technologies 2005, 2006). The fourth and final Study task will be a Final Report that summarizes and synthesizes the results of the field investigations, modeling applications, and the application and recommended future uses of the SacEFT.

The Study components are inter-related. For example, this SOS Report includes conceptual models that define functional relationships between ecosystem processes (e.g., bank erosion), habitats (e.g., vertical cutbanks), and the biotic needs of several species (e.g., nesting sites for bank swallows), and these functional relationships help form the structure of the SacEFT. This SOS Report also identifies and distills some of the available information and data for the Sacramento River to facilitate its incorporation into the SacEFT. Additionally, this SOS Report also defines conceptual models, key uncertainties and hypotheses that have helped to shape the focus of the field study plans and modeling applications. For example, hypotheses about the effects of changing gravel quality on salmonid spawning have shaped the methods and study sites used in the gravel field study. Similarly, the sediment transport model developed for this project has been used to test and shape hypotheses that are included in this SOS Report.

This version of the SOS Report is provided for public review and comment. A final version, incorporating appropriate changes, will be available in early 2007.

1.3 Other Planning Efforts in the Sacramento River

The overall goal for the Study is to provide information and tools that will be useful for other planning activities that are being conducted for the Sacramento River.

1.3.1 Water operations planning

Water operations planning in the Sacramento Valley considers a number of flow-related ecological targets, such as the maintenance of water temperatures to support winter-run Chinook salmon spawning in the upper Sacramento River, minimum in-stream flows, ramping rates to avoid fish stranding, the position of X2 (2 parts per thousand salinity) downstream in the Sacramento-San Joaquin Delta, and timing of pumping in the delta driven by considerations for fish species. However, there are few other ecological flow targets defined for more than two hundred miles of the mainstem river corridor, which prevents water managers from incorporating the full complement of Sacramento River ecosystem needs into water operations and flood management planning. At worst, the lack of this information can also lead to further degradation upstream on the Sacramento River system when ecological considerations are driven primarily by Delta-based targets. In contrast, ecological synergies may exist where a change in water management implemented to benefit the upper Sacramento System may also benefit targets in the Delta, yet these synergies remain to be investigated. Better definition of ecological flow needs can provide water operations planners and stakeholders with clearer targets to consider when exploring alternatives for water management in the Sacramento River. Creating an information base and a dialogue that includes ecological targets in this management may facilitate the creation and maintenance of more aquatic, floodplain, riparian, and off-channel habitat in the Sacramento River corridor, thereby improving habitat conditions for numerous species and potentially facilitating a reduction in the level of regulations that now dominate management of these species.

It is important to emphasize that this Study does not aim to return the Sacramento River flow patterns to some unaltered historical condition. Rather, the Study encourages resource managers and stakeholders to examine ways to manage water to better support the habitats and species of the Sacramento River while fulfilling water supply, water quality, and flood management objectives. In the long run, better maintenance of Sacramento River habitats and species can improve water supply reliability by enhancing populations of currently listed species and by helping to prevent the listing of additional species, thereby easing current restrictions on water supply operations or preventing the application of new restrictions.

1.3.2 Restoration planning

The Sacramento River corridor is the focus of considerable habitat restoration activity. For example, state and federal agencies have injected approximately 240,000 yds^3 (185,022 m^3) of spawning-sized gravel in the upper Sacramento River since 1978. CALFED and Central Valley Project Improvement Act (CVPIA) grants have supported the restoration and preservation of hundreds of acres of floodplain and riparian habitats in the middle Sacramento River. Tens of millions of dollars have been spent equipping water diversions of various sizes with fish screens, and new fish passage facilities at the Anderson Cottonwood Irrigation District (ACID) dam have improved salmonid access to upstream spawning habitats. This Sacramento River Ecological Flows Study strives to contribute to restoration planning for the Sacramento River by identifying the flow characteristics required to drive process-based restoration of aquatic, bank, riparian, and off-channel habitats, such as the routing of spawning-sized gravel, the formation of vertical cutbanks, and the recruitment of riparian vegetation on exposed surfaces. The Study also intends to identify high-priority, land based strategies that complement ecological flow needs in the Sacramento River, such as the continued injection of salmonid spawning-sized gravels, the removal of bank armor, and setting back levees.

1.3.3 Species recovery planning

Several species that occur in the Sacramento River are currently listed as endangered or threatened under the federal or California Endangered Species Act, including bank swallow (*Riparia riparia*), steelhead (*Oncorhynchus mykiss*), and two different runs of Chinook salmon (*Oncorhynchus tshawytscha*). NOAA National Marine Fisheries Service (NMFS) has also recently listed the southern population of North American green sturgeon (*Acipenser medirostris)*, which spawns in the Sacramento River, as threatened (NMFS 2006). State and federal agencies have prepared recovery plans for some of these listed species, such as bank swallow (CDFG 1992) and winter-run Chinook salmon (NMFS 1997), but recovery planning is still active in the Sacramento River basin. For example, NMFS has empanelled a Technical Recovery Team (TRT) to revise the recovery plan for Central Valley winter-run Chinook salmon evolutionarily significant unit (ESU), and to draft recovery plans for the Central Valley spring-run Chinook salmon and the Central Valley steelhead ESUs. This Study will provide information and tools (e.g., conceptual models, quantitative models, definition of research needs, the SacEFT) that can be used by the scientists and managers engaged in revising or drafting recovery plans for these species.

1.3.4 Current storage investigations

There are two significant water storage investigations being conducted in the Sacramento Valley: enlarging Shasta Dam/Reservoir (http://www.usbr.gov/mp/slwri/index.html) and a North-of-Delta-Offstream-Storage (NODOS) facility (a.k.a. Sites Reservoir) (http://www.usbr.gov/mp/nodos/index.html). Each storage option would likely change how flows are released, diverted, and routed in the Sacramento River, with attendant effects on habitats and species. This Study does not address directly either storage option, but by defining ecological flow needs, it will provide a series of targets that can be used by others to evaluate the potential effects (both positive and negative) of either storage option on Sacramento River processes, habitats, and species.

1.4 Study Area

The original proposal for this Study defined a study area between Red Bluff (RM 243) and Colusa (RM 143), where the channel has some freedom to migrate across the floodplain. However, the Study participants extended the upstream limit of the study area to Keswick Dam (RM 302), thereby adding another 59 river miles to the study area (Figure 1-1), to incorporate upstream habitats that are critically important for numerous species, including the listed focal species selected for this report. The study area focuses on the mainstem Sacramento River corridor, including the channel, adjacent floodplain and riparian habitats, and off-channel bodies. The Study does not address the entire Sacramento River basin or the tributaries of the river. Figures 1-2 and 1-3 provide additional details on the location of key features within the study area.

We realize that flows in the mainstem Sacramento River cannot be considered in a vacuum because of the complex water supply system and potential downstream effects of actions taken within the study area. Though the current Study does not attempt to formally integrate our study area results with other regions such as the Delta, we hope that the information produced through the project can be exported to other planning processes where it can be integrated by other experts. We also have made the Study flexible so that other issues, species, and regions can be addressed in future phases of the Study.

1.5 Study Approach

The processes, habitats, and species of the Sacramento River have been the focus of much study, and the volume of available reports and datasets poses a challenge for synthesizing information and organizing a discussion of ecosystem components. Divergent conceptual models about process-habitat-biotic linkages complicate the process of summarizing what is known about the Sacramento River, and add to the challenge of evaluating alternative approaches for conserving and restoring the river ecosystem. To help overcome these challenges, this SOS Report presents an overview of the Sacramento River through the lens of six focal species. A focal species approach facilitates the exploration of linkages among ecosystem processes, resultant habitats, and biotic needs. For each focal species, we identify the different life history stages that occur in the Sacramento River, the habitats used by each of those life history stages, the ecological processes that create and maintain those habitats, and the management actions (e.g., changes in the flow regime or bank protection) that influence those ecological processes and habitat conditions. The six focal species selected for this SOS Report are:

- Chinook salmon (*Oncorhynchus tshawytscha)*,
- steelhead (*Oncorhynchus mykiss*),
- green sturgeon (*Acipenser medirostros*),
- bank swallow (*Riparia riparia*),
- western pond turtle (*Clemmys marmorata*), and
- Fremont cottonwood (*Populus fremontii*).

Although this is by no means an exhaustive list of all conservation targets within the study area, the selection of these six focal species provided a logical starting point for the Study while covering a relatively wide range of habitats and ecological processes that occur in the Sacramento River. This section of the report first describes the process and criteria used to select these focal species.

1.5.1 Focal species selection

Stillwater Sciences has developed a set of criteria and a vetting process for selecting focal species, as illustrated in Figure 1-4**.** The methodical application of criteria to a pool of candidate species facilitates a comparison of the species, which clarifies and simplifies the process of selecting a suite of focal species.

1.5.1.1 Step 1: The species currently exists, or existed historically, within the target system

The first step of the vetting process involves determining if a candidate focal species currently exists, or existed historically, within the study reach. Species that currently occur in the system demonstrate an adaptation to current habitat conditions, so that the conservation and enhancement of existing habitat would likely not pose a threat to an existing population. This step also allows for the re-introduction of an extirpated species, which can be a goal of a restoration program.

Because many ecosystems currently support non-native species, the first step of the vetting process does not eliminate non-native species from consideration as a focal species. Non-native species can serve as valuable focal species, especially if they are strong interactors in the system, by clarifying or increasing our knowledge of the environmental changes that have conferred a competitive advantage to them. Such knowledge can assist the design of management actions that reduce that competitive advantage. Though it is often infeasible to eradicate a non-native species once it has become widely established, management actions may help to control the abundance or distribution of targeted non-native species so that their ecological effects are reduced.

1.5.1.2 Step 2: Is the species listed as endangered or threatened?

The second step of the vetting process acknowledges that the recovery of listed species constitutes a high social priority, both economically and ecologically. It also recognizes that listed species are often at the center of resource management conflicts, so that recovery of the species can be an important management goal as a means of reducing conflict with, and restrictions on, human activities. The endangered and threatened species that occur in an ecosystem often serve as focal species; however, the number of listed species that occur in the Sacramento River corridor generally precludes the selection of every listed species as a focal species. One of the functions of the focal species approach is to facilitate the synthesis, analysis, and organization of information by engaging a manageable number of species; however, this process can be undermined by the selection of too many focal species.

1.5.1.3 Step 3: Additional criteria for non-listed species

The third step of the selection process provides much of the information used to compare candidate focal species by applying a series of criteria to non-listed species. It is often important to include non-listed species in the group of focal species in order to capture potential ecosystem changes that are reducing their populations, which could necessitate future protection that would exacerbate resource conflicts.

- **Other special-status designation**. The first criterion queries whether an unlisted species has some other special-status designation (e.g., species of concern). For example, NMFS has designated both Central Valley fall-run and late fall-run Chinook salmon as species of concern because of recent population trends, indicating that further reductions in escapements could necessitate future listing and protection (NMFS 2004).
- **High economic or public interest value**. The second criterion recognizes the economic or social importance of certain species, such as species that are the focus of commercial fisheries (e.g., salmon) and sportfish that are the focus of recreational angling (e.g., steelhead, sturgeon).
- **Narrow habitat requirements**. The third criterion tests whether a species has narrow habitat requirements such that loss of that habitat type would pose a significant threat to the health of the population. For example, bank swallows nest in fresh vertical cutbanks composed of soils with a loamy-sandy texture and at least 1 m in height, which represents a stringent mix of habitat conditions. Bank swallow colonies also have a limited lifespan (< 7 years) because of bank slumping, rodent burrowing, and possibly parasite infestation. Consequently, activities that affect the frequency of bank erosion in zones of appropriately textured soils (e.g., bank protection, flow regulation, land conversion) can combine with the narrow habitat requirements of bank swallow to create a significant threat to nesting.
- **Weak disperser**. The fourth criterion identifies species that have difficulty dispersing to new areas, which prevents a species from establishing new sub-populations that can help mitigate the loss of an existing breeding/spawning population from a catastrophic event. For example, even though green sturgeon migrate thousands of miles through rivers, estuaries, and ocean, there are only three known spawning populations of green sturgeon, which suggests that the species has difficulty establishing new spawning sub-populations outside of the current populations in the Sacramento, Rogue, and Klamath rivers. As a consequence, a natural or anthropogenic event that eliminates habitat in one of these three river systems could dramatically reduce the range of the species.
- **Strong Interactor**. The fifth criterion indicates that particular species can significantly influence natural communities through ecological interactions with other species. For example, a species may serve as an important prey species for a number of other species, such that a decline in its population can reduce the food base for other species and depress the abundance of an entire community. Similarly, other species can affect a community by monopolizing available habitat and resources or by preying on a wide variety of species (e.g., the threat posed by an introduction of northern pike (*Esox lucius*) in Central Valley rivers). Other species can change the very nature of an ecosystem (e.g., Asian clam (*Potamocorbula amurensis*) converting portions of the Bay-Delta estuary from a pelagic to a benthic system).
- **Loss of habitat**. The sixth criterion addresses a factor that often contributes to reductions in the abundance and/or distribution of a species—habitat loss and degradation as a function of anthropogenic changes to the system. For example, all salmonids in the Central Valley have experienced dramatic losses of spawning habitat as a function of large water supply dams that have eliminated access to historical spawning grounds. This criterion highlights that changes in the current management of resources (e.g., flow, gravel, large woody debris [LWD], available floodplain) have the potential to improve ecosystem conditions that support species, often in spite of historical losses of habitat and habitat quality.

- **Local and/or regional population declines**. The final criterion acknowledges that population abundance and distribution provide two of the key metrics for assessing the health of a species. Local and regional population declines provide a warning signal that a system is undergoing change, thus providing a stimulus for identifying the factors affecting a population. Continued population declines can also necessitate eventual protection under the Endangered Species Act, which often intensifies conflicts over natural resources.

1.5.1.4 Step 4: Availability of information

If a species satisfies at least one of the criteria identified in Step 3 of the vetting process, then it passes to Step 4, in which the information about a species is assessed. At a minimum, we must know the general habitat requirements and life history stages of a species for it to function as a focal species. Although it is preferable if this information is specific to the Sacramento River study system, knowledge of how a species interacts with its environment in a similar system is also of value. For example, although there is little specific information about the abundance and distribution of western pond turtle in the Sacramento River system; research conducted in other river systems provides useful information about general habitat preferences that we can apply for analysis in the Sacramento River. The more detailed knowledge that we have of a species, then the greater utility that species can provide as a focal species. Ideally, we will have quantitative data about the abundance, distribution, and habitat preferences of a species. For example, several studies have identified the general range of preferred flow velocities, flow depths, and water temperatures of spawning Chinook salmon (Bovee 1978, Burner 1951, Chapman et al. 1986, McCullough 1999, Smith 1973), which have been applied in previous studies of salmon in the Sacramento River (USFWS 2003, 2005).

1.5.1.5 Step 5: Ranking of species

The information produced for each candidate species in Steps 2, 3 and 4 facilitates a general ranking of species in Step 5 of the vetting process. Rankings can be either nominal (e.g., high, medium, low priority) or ordinal (e.g., 1st, 2nd, 3rd, etc.). To select focal species for the SOS Report, we used nominal rankings. Species receiving high rankings needed to have adequate information available (Step 4) and had to be officially listed (Step 2) or meet two or more criteria listed under Step 3.

1.5.1.6 Step 6: Select focal species

The rankings from Step 5 are used to inform the final selection process in Step 6. Selection of the final suite of focal species can include species that, at least partially, represent different assemblages or guilds and species that utilize a broad range of habitat types within the study reach, so that the synthesis and analysis of information will be relevant to a broad range of species.

Selecting too many focal species can undermine the purpose of a focal species approach, which is to focus and organize the discussion and analysis in a manner that is still relevant to a broad array of species. Early in this project, we determined that Chinook salmon was likely to be selected as a focal species. Because this focal species covers four separate runs (e.g., winter-run, spring-run, fall-run, late-fall-run), we recognized that selecting Chinook salmon as a focal species was akin to selecting four different focal species in light of the volume of available information and the work required to synthesize and analyze that information for each run. As a result, we determined that a total of five or six species would constitute a manageable suite of focal species that would cover a broad range of habitat types that occur in the river corridor. If two or more candidate species used similar habitat types, the one with the highest ranking in Step 5 was selected.

For this Study the study team adapted the vetting process by selecting a pool of ten candidate focal species that we hypothesized were highly responsive to changes in the Sacramento River's flow regime. We also identified species that are at the center of resource management conflicts or the object of significant study in the basin. The pool of candidate species included:

- Chinook salmon
- steelhead
- green sturgeon
- bank swallow
- western pond turtle
- Fremont cottonwood
- Sacramento splittail
- yellow-billed cuckoo
- Swainson's hawk
- largemouth bass

The following sections describe the vetting process used for each candidate species to explain its inclusion or exclusion from the final group of focal species.

1.6 Candidate Focal Species

1.6.1 Chinook salmon (*Oncorhynchus tshawytscha*)

The Sacramento River is unique because it supports four distinct runs of Chinook salmon: winter-run, spring-run, fall-run, and late fall-run. Each of these runs occurred historically in the mainstem Sacramento River (Step 1). Two of the runs are protected by the federal and the California Endangered Species Acts (CESA): winter-run (endangered) and spring-run (threatened) (Step 2). The two remaining runs (fall-run and late fall-run) have been designated as species of concern by NMFS (Step 3a). All of the salmon runs have high economic value because they support commercial fisheries (Step 3b). Each run also has high public interest value, both as a charismatic megafauna that appeals to the broad public and as a target of recreational angling (Step 3). Numerous human activities have reduced the extent and quality of habitats that Chinook salmon used historically (Step 3). Water supply and diversion dams impede access to historical spawning grounds. Large dams also trap sediment from the upper watershed, depriving downstream reaches of a fundamental building block of salmon habitat. Bank protection and land use changes have reduced the recruitment of LWD to the channel, which has reduced velocity refugia and rearing habitat for juvenile salmon. Bank protection has altered channel geometry and geomorphic features (e.g., point bars) by stimulating channel narrowing and further reducing sediment supply, with attendant effects on salmon spawning and rearing habitat. Each salmon run in the Sacramento River has also experienced general population declines in the last 35 years (Step 3), which has stimulated numerous restoration and recovery efforts (e.g., Anadromous Fish Restoration Program [AFRP], CALFED). There is a significant volume of information available for the different runs of Chinook salmon that occur in the Sacramento River because salmon have been the primary object of study in the basin (Step 4). All of these factors combine to produce a high priority ranking for Chinook salmon (Step 5) and to explain their selection as a focal species for this SOS Report (Step 6).

1.6.2 Steelhead (*Oncorhynchus mykiss*)

Many of the factors described for Chinook salmon also apply to steelhead. Steelhead occurred historically in the Sacramento River basin, often ascending to high elevation reaches of tributaries to the Sacramento River (Step 1) (Yoshiyama et al. 1996). Steelhead is also a federally listed threatened species (Step 2). Steelhead generate high public interest because they are prized targets of recreational anglers, and they appeal to the broader public as a charismatic megafauna associated with wild places and California history (Step 3). As with Chinook salmon, steelhead have been displaced by major water supply dams that impede access to the majority of historical spawning habitat (Step 3). Dams have also eliminated access to the vast majority of historical rearing habitat for steelhead, which used to rear for two years in high elevation reaches where year round water temperatures were cold so that juveniles could survive through the warm summers. The loss of habitat has stimulated declines in the abundance of steelhead (Step 3), which has necessitated their listing as a threatened species. Though steelhead stocks throughout the Pacific Northwest have been the object of much study, we know relatively little about the specific habitat preferences of the steelhead population that spawns in the mainstem Sacramento River. Nevertheless, we can use information derived from other sub-populations to understand the general habitat requirements of steelhead in the Sacramento River (Step 4).

Steelhead received a high priority ranking because they are a listed species, they satisfied multiple criteria in the third step of the vetting process, and we know enough about their general life history stages and habitat requirements to understand how changes in the system may affect them. Although Chinook salmon and steelhead are often grouped together because of an assumption that management and restoration targeted at salmon will also benefit steelhead, steelhead exhibit sufficiently different life history timing and strategy from the Chinook salmon runs to merit inclusion as a separate focal species (Step 6). In addition to differences in run timing with Chinook salmon, steelhead rear in the river for two years before smolting, which can present unique issues for resource management, especially the use of flows to manage water temperatures.

1.6.3 Green sturgeon (*Acipenser medirostros*)

The Sacramento River supports one of only three known spawning populations of green sturgeon (Step 1), which was recently listed as a threatened species under the federal Endangered Species Act (Step 2) (NMFS 2006). Though green sturgeon migrate thousands of miles through rivers, estuaries, and ocean during their long lives, they do not appear to readily establish new sub-populations, which suggests that they are a weak disperser (Step 3). As a result, threats to any one of the three spawning populations could significantly reduce the range of the species. The construction of Keswick and Shasta dams has likely impeded access to all of the spawning habitat that green sturgeon used historically (Step 3). Green sturgeon can spawn in a wide range of environments in terms of flow velocities, flow depths, and particle size of the channel bed; however, they are reported to prefer areas of fast, deep, turbulent water associated with slope breaks in the channel (Moyle 2002). Based on the general location of green sturgeon spawning in the Rogue and Klamath river systems, it is likely that such preferred spawning habitat occurred historically upstream of Shasta Dam. The population of green sturgeon that spawns in the Sacramento River seems to have experienced a recent population decline (Step 3), which stimulated the petition for federal protection under the Endangered Species Act and its current status as a threatened species. Though we have a broad understanding of green sturgeon spawning habitat requirements, investigators have not yet identified specific spawning locations in the Sacramento River, which limits the amount of detailed information available for the species (Step 4). Current radio telemetry studies of green sturgeon will likely contribute to our understanding of green sturgeon spawning in the Sacramento River in the next few years (J. Heublein, pers. comm., 2005). Recent laboratory research has identified water temperature thresholds for larval green sturgeon (Van Eenennaam 2001); however, little else is known about specific rearing locations or rearing habitat preferences in the Sacramento River.

Despite the lack of detailed knowledge about specific green sturgeon spawning locations and rearing habitat preferences in the Sacramento River, we have a general understanding of the life history stages that occur in the study reach and the general habitat requirements of the species. The status reviews conducted by NMFS (Adams et al. 2002, NMFS 2005) also provide a useful compendium of available information for the species. Green sturgeon life history timing and habitat needs are also sufficiently different from the salmonids to prevent unnecessary overlap with other selected focal species. Many of the factors that have stimulated NMFS to list Sacramento River green sturgeon as a threatened species (e.g., recent population declines, loss of historical spawning habitat caused by human activities, one of only three known spawning populations) also compel the inclusion of green sturgeon as a focal species for this SOS Report (Step 6).

1.6.4 Bank swallow (*Riparia riparia*)

The Sacramento River supports one of the few nesting populations of bank swallows in California (Step 1). Bank swallows are listed as a threatened species under CESA (Step 2). They have narrow habitat requirements for nesting, because they need tall (> 1 m [> 3 ft]) vertical cutbanks located in friable soils (Step 3). The steepness and height of cutbanks helps to prevent predators from accessing the nesting burrows, and friable soils are required to allow burrowing. Because vertical cutbanks generally slump over time as flow events erode the toes of the banks, fresh bank erosion of loamy-sandy soils is required to provide new colony sites. The Sacramento River nesting population has experienced a general decline in abundance (Step 3) which contributed to its listing as a threatened species.

The state of California drafted a recovery plan for bank swallow, which provides a good resource for understanding the habitat needs of the species (Step 4), and recent research has advanced our knowledge of their habitat requirements (Moffatt et al. 2005). Several factors combine to merit a high priority ranking for bank swallow (Step 5), including its status as a protected species, its narrow habitat requirements, the availability of information about its habitat needs, and the availability of recent survey data on the abundance and distribution of the species within the Sacramento River. As a result, bank swallow was selected as a focal species for the SOS Report (Step 6).

1.6.5 Western pond turtle (*Clemmys marmorata*)

The current range of western pond turtle includes the Sacramento River corridor, though the current distribution of the species in the basin has likely been reduced from its historical distribution (Step 1). Though the western pond turtle is not currently listed as an endangered or threatened species (Step 2), it has been designated as a species of concern by both federal agencies and the state of California (Step 3). Western pond turtle habitats have likely been reduced in extent and quality from historical conditions as a function of land use changes that have converted habitat to agriculture and urban development (Step 3). Dam construction and operations have also altered the flow and sediment regime, which have likely combined with levee construction and bank protection activities to reduce bank erosion and meander migration, which may affect the formation of off-channel habitats that appear to provide the majority of the aquatic habitat for western pond turtle in the Sacramento River corridor. The general abundance and distribution of western pond turtle has also been shrinking throughout their range (Step 3), which has contributed to its designation as a species of special concern in California.

Though western pond turtles are known to occur in the Sacramento River, there is relatively little information about their distribution within the corridor. Nevertheless, research conducted on other rivers provides a general understanding of their life history stages and habitat requirements that can guide inquiry in the Sacramento River (Step 4). The general habitat requirements and preferences of western pond turtle also provide a linkage with a range of off-channel habitat types (e.g., oxbow lakes, sloughs, side channels) that are not well-covered by other candidate focal species, which was a key factor in selecting the western pond turtle as a focal species (Step 6). Further, the distribution and abundance of these off-channel habitats are strongly linked to management actions being evaluated by this Study, including flow regime, and levee and riprap alignment. In addition, the western pond turtles were also unique among the pool of candidate species because they use both aquatic and terrestrial habitats.

1.6.6 Fremont cottonwood (*Populus fremontii*)

Historically, the Sacramento River was flanked by miles of riparian forest, including large stands of Fremont cottonwood (Step 1). Cottonwood is not listed as threatened or endangered (Step 2), but it is often used as an indicator species for riparian vegetation communities in the Central Valley, which provide habitat for a variety of special-status species (Step 3). Cottonwood can be considered to have narrow habitat requirements (Step 3), because it depends on the availability of bare mineral soils and periodic flooding during seed release periods to colonize new areas. It is also a strong interactor in aquatic and riparian communities by providing habitat for numerous species, by supplying LWD to the channel for habitat complexity, and by helping to stabilize banks (Step 3). Cottonwood forest has been dramatically reduced from its historical extent through clearing for fuel, agricultural conversion of habitat, and flow regulation by water supply operations (Step 3). There is considerable information about cottonwood recruitment dynamics, including recent research in the Central Valley and on the Sacramento River in particular (Step 4). Though cottonwood is not a listed species, it satisfied several of the criteria in Step 3 of the vetting process. Because it is generally used as an indicator species for riparian vegetation communities, and because it provides important habitat for numerous species, cottonwood received a high priority ranking (Step 5) and was selected as a focal species for the SOS Report (Step 6).

1.6.7 Sacramento splittail (*Pogonichthys macrolepidotus*)

Splittail spend the majority of their life in the Bay-Delta estuary, but they migrate up the Sacramento River to spawn, with some adults ascending as far upstream as Hamilton City (RM 200) and Red Bluff Diversion Dam (RM 243), as evidenced by the capture of splittail in rotary screw traps (Sommer et al. 1997). However, the bulk of splittail occurrence in the Sacramento River basin is downstream of the confluence with the Feather River (Moyle et al. 2004), which falls outside the study area defined for the Study (Step 1). Though splittail is not currently listed as endangered or threatened (Step 2), it was previously listed as a federal threatened species between 1999 and 2003, and it is currently a California species of special concern (Step 3) (Moyle et al. 2004). Splittail has little value as a recreational sportfish, so it generally has low economic and public interest value (Step 3). Though the distribution of splittail has likely been reduced in the Sacramento basin as a function of habitat loss caused by human activity, it is unclear if splittail were ever abundant in the upper Sacramento River historically (Step 3). It is also difficult to discern a clear historical pattern in the abundance of splittail, and therefore whether their population is declining (Step 3), because they have a wide range of natural variability, as illustrated by the listing and then the de-listing of splittail as a federal threatened species.

Splittail were ranked a low priority (Step 5), primarily because they are not abundant within the study reach defined for the Study of which the SOS Report is a part (Step 1), but also because they did not clearly satisfy many of the criteria applied in Step 3 of the vetting process. As a result of this low ranking, it was not selected as a focal species. However, for future projects that include habitats downstream of Colusa (RM 143), particularly floodplain habitat, it might be worthwhile to consider splittail as a candidate focal species

1.6.8 Western yellow-billed cuckoo (*Coccyzus americanus occidentalis*)

The Sacramento River supports the largest number of nesting pairs of yellow-billed cuckoo in California (Step 1). Western yellow-billed cuckoo was listed as an endangered species by the state of California in 1988 (Step 2). Yellow-billed cuckoo depend on thick patches of riparian vegetation, which has been reduced by human activity (e.g., agricultural conversion of habitat) (Step 3). Regional population declines stimulated the need for listing yellow-billed cuckoo as an endangered species (Step 3). Recent research and monitoring have contributed to a reasonable understanding of the habitat requirements and current distribution of the species (Step 4), although much is still unknown. Though yellow-billed cuckoo received a high priority ranking (Step 5), it was not selected as a focal species for the SOS Report (Step 6) because its Sacramento River habitat needs could be partially represented by the selection of Fremont cottonwood as a focal species. The chapter on Fremont cottonwood contains a section on riparian songbirds (Section 9.4), which includes western yellow-billed cuckoo. In addition, Fremont cottonwood is preferable as a focal species because it has more direct and better understood links to the flow regime.

1.6.9 Swainson's hawk (*Buteo swainsoni*)

Most nesting pairs of Swainson's hawk are found downstream of the study area for the Study, in the vicinity of Sacramento and the Delta, though nesting pairs have been found within the study area (Step 1). Swainson's hawk was listed as a threatened species by the state of California in 1983 (Step 2). Swainson's hawk requires vegetation associated with the riparian corridor, often in proximity to open fields, so human activities have likely reduced habitat extent and quality (Step 3). There is enough information about Swainson's hawk to support the identification of life history stages and general habitat needs in the Central Valley (Step 4). Because the majority of known nesting pairs are located outside of the study area, Swainson's hawk received a low priority ranking (Step 5), so it was not selected as a focal species for this SOS Report (Step 6). In addition, this species does not have strong linkages to the primary management actions being considered in this Study.

1.6.10 Largemouth bass (*Micropterus salmoides*)

Largemouth bass is a non-native species that currently occurs in the Sacramento River system (Step 1). As an alien species, it is not listed as threatened or endangered within the Central Valley (Step 2). Though it does not support any large commercial enterprise, largemouth bass generates some public interest as a target of recreational anglers (Step 3). Largemouth bass may be considered a moderately strong interactor in aquatic habitats because of its predation effects on other native species such as juvenile Chinook salmon and possibly western pond turtle hatchlings (Step 3).

The habitat requirements and life history timing of largemouth bass are fairly well understood, although there is little specific information about the distribution and abundance of the species in the Sacramento River corridor (Step 4). Largemouth bass received a low ranking (Step 5), primarily because the species only satisfies a couple of the criteria defined in Step 3 of the vetting process. As a result, largemouth bass is not included as a focal species in the SOS Report.

1.7 SOS Report Organization

This report contains ten chapters. This introductory chapter describes the approach used for compiling and structuring this SOS Report, including the focal species approach used to focus Study inquiry. Chapter 2 describes a timeline of the key anthropogenic changes to the mainstem Sacramento River corridor, followed by a summary of existing information about how those key changes affected the river's hydrology, geomorphology, habitats, and biota. Chapter 3 describes the hydrological and fluvial geomorphic processes that create and maintain key habitats for the six focal species. Chapter 4 is the first of several focal species chapters. Each focal species chapter describes:

- the status of the focal species
- the historical distribution and life history timing of the focal species
- the effects of key anthropogenic changes on the focal species
- the current life history timing and habitat needs of the focal species
- key flow and habitat actions to consider to improve habitat extent and quality for the focal species
- key hypotheses and uncertainties that need to be addressed through future research.

Chapters 4 through 6 address fish species, so they focus primarily on aquatic habitats. Chapter 4 addresses the four runs of Chinook salmon (*Oncorhynchus tshawytscha)* in separate subsections: Section 4.1 provides an overview of Chinook salmon life history, Section 4.2 addresses winter-run Chinook salmon, Section 4.3 discusses spring-run Chinook salmon, Section 4.4 covers fall-run Chinook salmon, and Section 4.5 addresses late-fall-run Chinook salmon. Chapter 5 analyzes the steelhead (*Oncorhynchus mykiss)* population that spawns in the mainstem Sacramento River, and Chapter 6 addresses North American green sturgeon (*Acipenser medirostris)*, which were recently listed as a threatened species. Chapter 7 discusses bank swallow (*Riparia riparia*), thereby addressing bank and terrestrial habitats. Chapter 8 presents the focal species chapter for western pond turtle (*Clemmys marmorata*), which incorporates a discussion of floodplain and off-channel habitats. Chapter 9 uses Fremont cottonwood (*Populus fremontii*) as an indicator species for riparian habitats, and includes a discussion of the songbird assemblage that is supported by riparian vegetation. Chapter 10 distills and integrates some of the key management recommendations that are defined in the focal species chapters.

1.8 References

Adams, P. B., C. B. Grimes, J. E. Hightower, S. T. Lindley, and M. L. Moser. 2002. Status review for North American green sturgeon, *Acipenser medirostris*. National Marine Fisheries Service, Santa Cruz, California.

Bovee, K. D. 1978. Probability-of-use criteria for the family Salmonidae. US Fish and Wildlife Service, Instream Flow Group Information Paper 12, Fort Collins, Colorado.

Burner, C. J. 1951. Characteristics of spawning nests of Columbia River salmon. Fishery Bulletin 52:95–110.

CDFG (California Department of Fish and Game). 1992. Recovery plan: bank swallow. Report No. 93.02. CDFG, Nongame Bird and Mammal Section, Wildlife Management Division, Sacramento.

Chapman, D. W., D. E. Weitkamp, T. L. Welsh, M. B. Dell, and T. H. Schadt. 1986. Effects of river flow on the distribution of Chinook salmon redds. Transactions of the American Fisheries Society 115:537–547.

ESSA Technologies Ltd. 2005. Sacramento River Decision Analysis Tool: workshop backgrounder. Prepared by ESSA Technologies Ltd., Vancouver, British Columbia for The Nature Conservancy, Chico, California.

ESSA Technologies Ltd. 2006. Sacramento River Ecological Flows Tool (SacEFT): design guidelines. Prepared for by ESSA Technologies Ltd., Vancouver, British Columbia for The Nature Conservancy, Chico, California.

Kondolf, G. M., T. Griggs, E. W.Larsen, S. McBain, M.Tompkins, J. G. Williams, and J. Vick. 2000. Flow regime requirements for habitat restoration along the Sacramento River between Colusa and Red Bluff. CALFED Bay Delta Program, Integrated Storage Investigation, Sacramento, California.

McCullough, D. A. 1999. A review and synthesis of effects of alterations to the water temperature regime on freshwater life stages of salmonids, with special reference to Chinook salmon. EPA 910-R-99-010. Prepared for U.S. Environmental Protection Agency Region 10, Seattle, Washington. www.critfc.org/tech/EPAreport.htm.

Moffatt, K. C., E. E. Crone, K. D. Holl, R. W. Schlorff, and B. A. Garrison. 2005. Importance of hydrologic and landscape heterogeneity for restoring bank swallow (*Riparia riparia*) colonies along the Sacramento River, California. Restoration Ecology 13: 391–402.

Moyle, P. B. 2002. Inland fishes of California. Revised edition. University of California Press, Berkeley.

Moyle, P. B., R. D. Baxter, T. Sommer, T. C. Foin, and S. A. Matern. 2004. Biology and population dynamics of Sacramento splittail (*Pogonichthys macrolepidotus*) in the San Francisco Estuary: a review. San Francisco Estuary and Watershed Science [online serial] 2: Article 3. http://repositories/cdlib.org/jmie/sfews/vol2/iss2/art3

NMFS (National Marine Fisheries Service). 1997. NMFS Proposed recovery plan for the Sacramento River winter-run Chinook salmon. NMFS, Southwest Region, Long Beach, California.

NMFS. 2004. Endangered and threatened species: proposed listing determinations for 27 ESUs of west coast salmonids. Federal Register 69: 33102–33179.

NMFS. 2005. Green sturgeon (*Acipenser medirostrus*) status review update. NMFS, Biological Review Team, Santa Cruz Laboratory, Southwest Fisheries Science Center.

NMFS. 2006. Endangered and threatened wildlife and plants: threatened status for Southern Distinct Population Segment of North American green sturgeon: final rule. Federal Register 71: 17757-17766.

Smith, A. 1973. Development and application of spawning velocity and depth criteria for Oregon salmonids. Transactions of the American Fisheries Society 102:312–316.

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

USFWS (U.S. Fish and Wildlife Service). 2003. Flow-habitat relationships for steelhead and fall, late-fall and winter-run Chinook salmon spawning in the Sacramento River between Keswick Dam and Battle Creek. Final report. USFWS, Sacramento Fish and Wildlife Office, Sacramento, California.

USFWS. 2005. Flow-habitat relationships for fall-run Chinook salmon spawning in the Sacramento River between Battle Creek and Deer Creek. USFWS, Sacramento Fish and Wildlife Office, Sacramento, California.

Van Eenennaam, J. P., M. A. H. Webb, X. Deng, S. I. Doroshov, R. B. Mayfield, J. J. Cech Jr., D. C. Hillemeier, and T. E. Willson. 2001. Artificial spawning and larval rearing of Klamath River green sturgeon. Transactions of the American Fisheries Society 130: 159–165.

Watercourse Engineering, Inc. 2002. Historic Flow and Temperature Modeling of the Sacramento River – Period of Simulation: 1970-2001. Prepared for the United States Geological Survey, Biological Resources Division, Mid-Continent Ecological Science Center, Ft. Collins, CO. March 28.

Yoshiyama, R. M., E. R. Gerstung, F. W. Fisher, and P. B. Moyle. 1996. Historical and present distribution of Chinook salmon in the Central Valley drainage of California. Pages 309–362 in Sierra Nevada Ecosystem Project: final report to congress. Volume III: Assessments, commissioned reports, and background information. University of California, Center for Water and Wildland Resources, Davis.

Figures

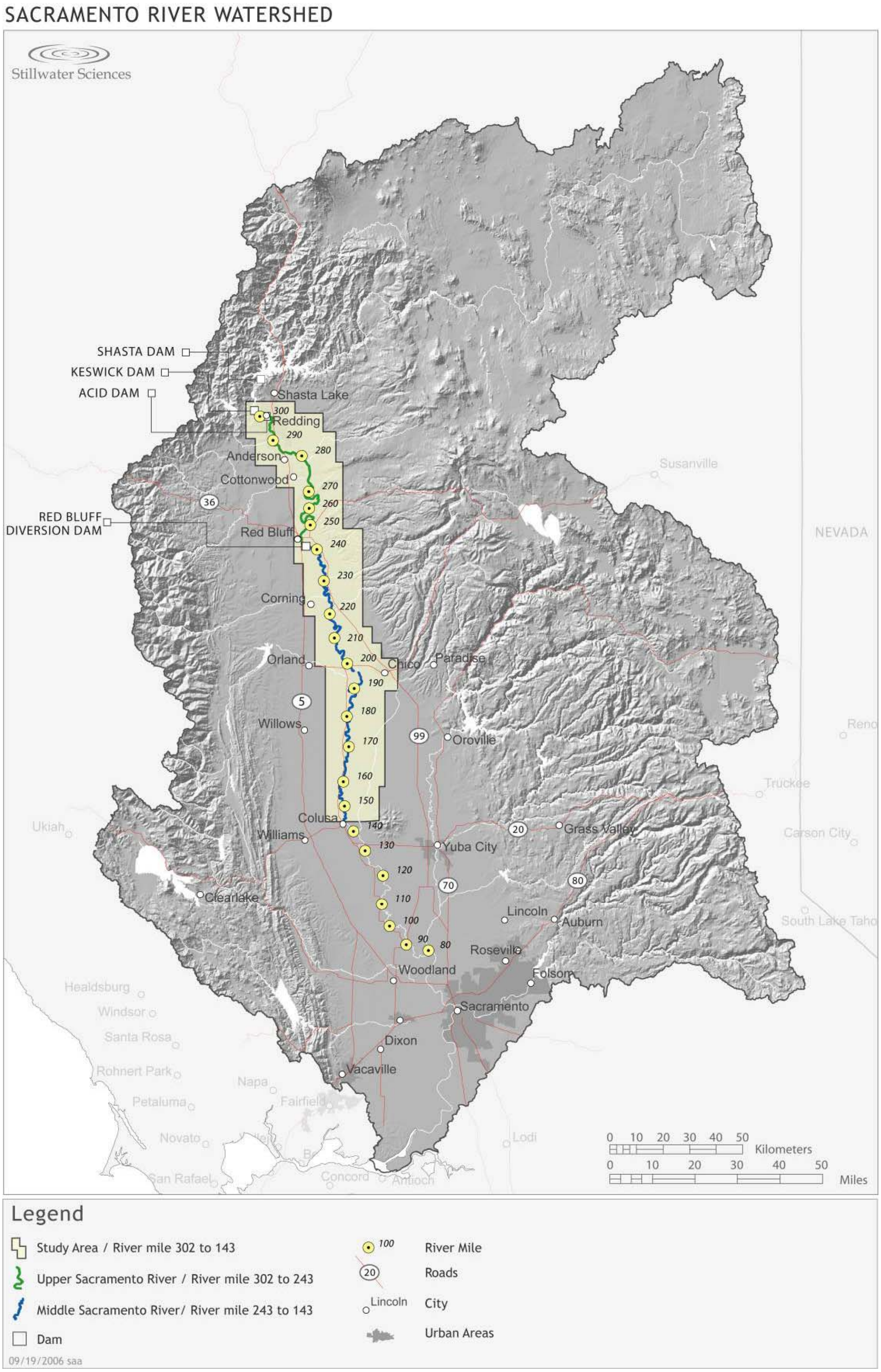


Figure 1-1. Sacramento River Watershed.

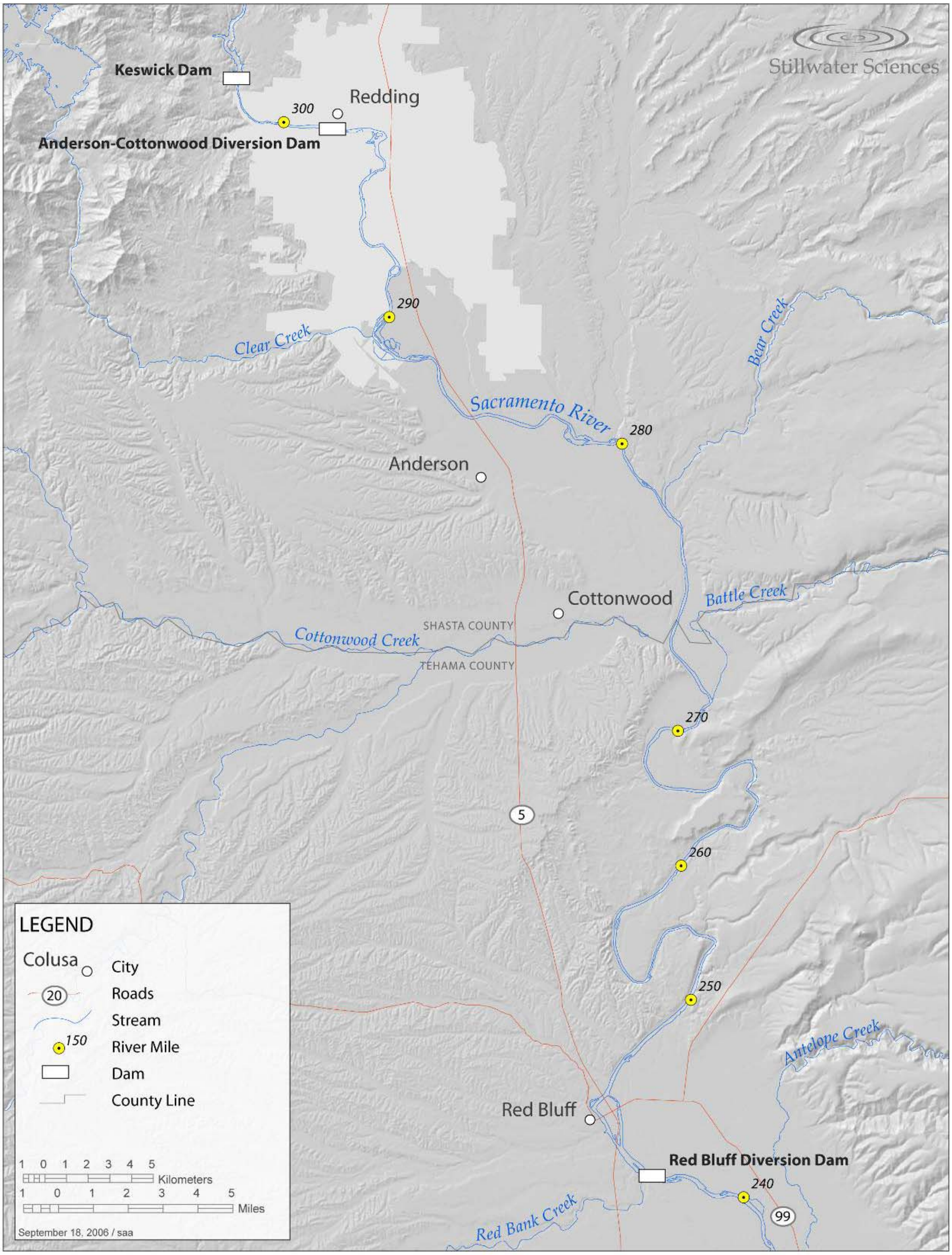


Figure 1-2. Upper Sacramento River, Keswick Dam to Red Bluff Diversion Dam.

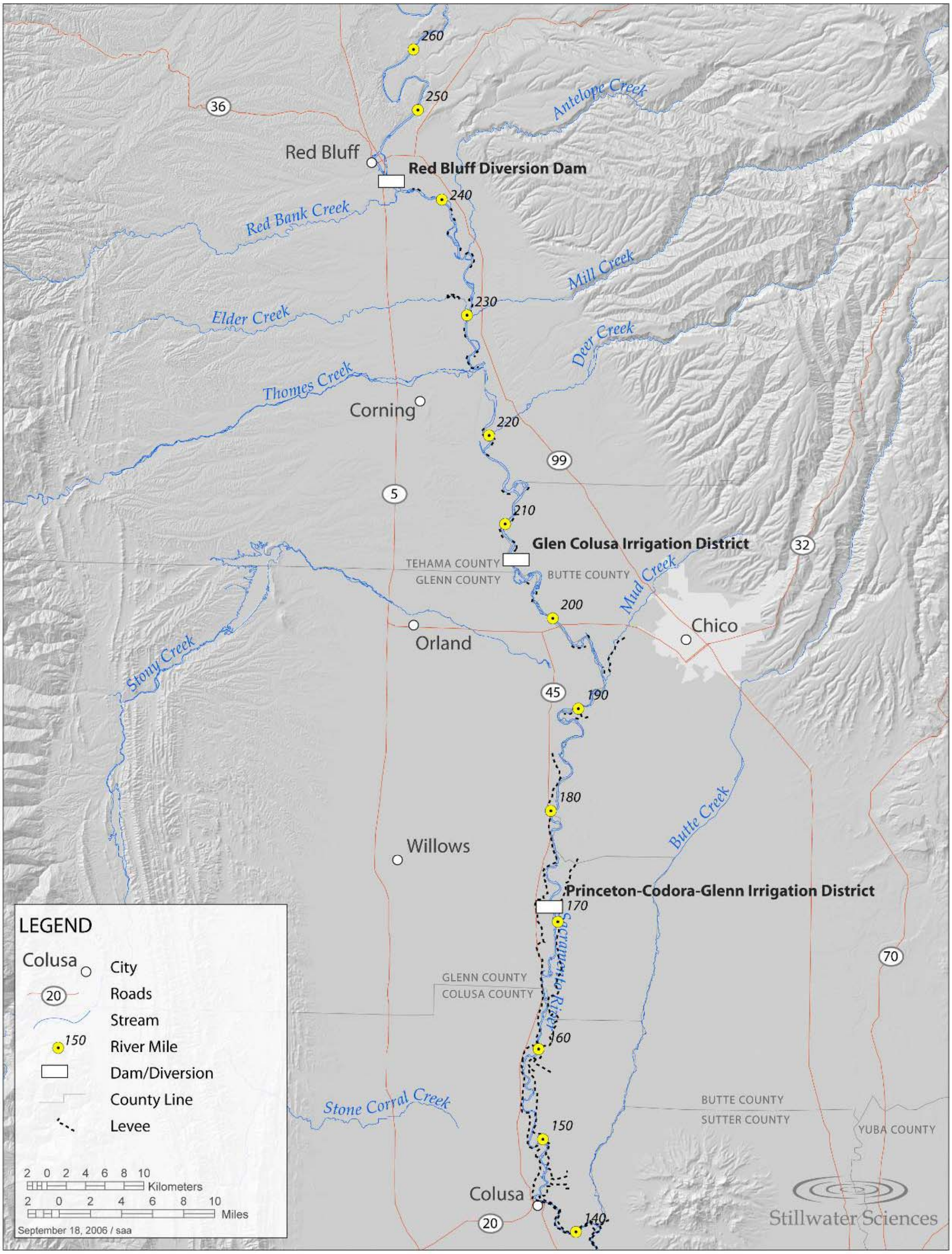


Figure 1-3. Middle Sacramento River, Red Bluff Diversion Dam to Colusa.

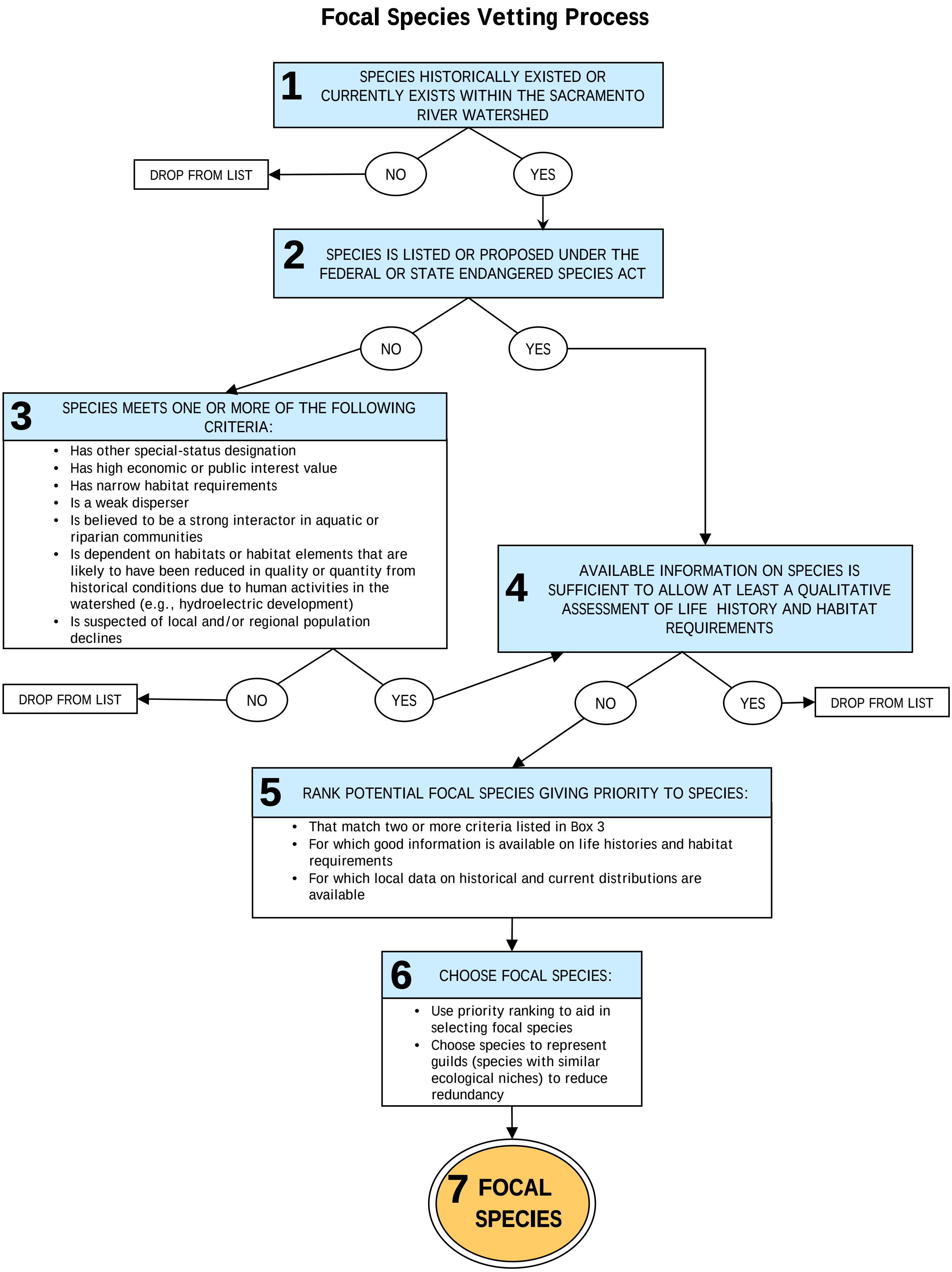


Figure 1-4. Focal Species Vetting Process.

2 HISTORICAL CONTEXT

This chapter provides an annotated timeline of significant human-induced changes to the Sacramento River and its floodplain. More detailed descriptions and discussion of individual events and changes will be presented in subsequent chapters, with emphasis on how human activities have affected geomorphic processes (Chapter 3) and the evolution of focal species habitats (Chapters 4 through 9).

The Sacramento River system has been the focus of extensive resource development over the past 150 years. Understanding how human activities have changed, and continue to change, the watershed inputs and ecological processes of the Sacramento River system provides context for understanding current resource conflicts and for assessing how human and ecosystem needs can be balanced through informed management decision making.

2.1 Anthropogenic Changes

The Sacramento River and its floodplain have provided much of the resources used to build California. In the late 19th century a robust agrarian economy developed on the river's fertile floodplain and eventually eclipsed gold mining as a key economic engine for the state. Development of the river's water supply helped fuel the growth of agriculture in the Sacramento and San Joaquin valleys. Sacramento River water also permitted nearly unchecked population expansion in a semi-arid environment that would have otherwise limited human settlement in California. The abundant fish and wildlife resources helped support population growth in the late nineteenth century, and fueled commercial activity in the early twentieth century. Riparian woodlands along the Sacramento River provided timber and fuel to support early human habitation, and the sand and gravel of the channel and floodplains provided aggregate for growth of urban centers and infrastructure.

Development of the basin's resources came at a price, however, as aquatic, floodplain, and riparian habitats were destroyed or degraded and populations of many native species plummeted. Figure 2-1 presents a timeline of significant anthropogenic changes to the Sacramento River corridor. Brief descriptions of some of the key human changes to the Sacramento River system are provided below.

2.1.1 Land conversion

Prior to Indo-European colonization, approximately 500,000 ac (200,000 ha) of riparian and upland forest flanked the Sacramento River in swaths as wide as 5 mi (8 km) (The Resources Agency 1989). These dense bands of vegetation provided vast habitat and nearly continuous migration corridors for many wildlife species. Removal of riparian vegetation was one of the earliest effects of increasing human habitation in the Sacramento Valley, as settlers cleared space for agriculture, grazing and homesteads beginning in the mid-nineteenth century. The Gold Rush in 1849 accelerated vegetation removal, as prospectors harvested wood for flume works and burgeoning urban centers increased the demand for lumber and firewood. The spike in population also stimulated the conversion of more land to agriculture and grazing to supply food. Steamships plied the Sacramento River up to Red Bluff, and crews often harvested local wood from the margins of the channel for fuel (The Bay Institute 1998). Overall, riparian and upland forests of the Central Valley have been reduced by nearly 95% over the past 150 years (The

Resources Agency 1989, The Bay Institute 1998). In addition to the clearing of riparian forest, the conversion of fertile floodplain land to agricultural uses often included filling off-channel water bodies.

In addition to the riparian and upland forest, much of the Sacramento valley floor was covered by permanent and seasonal wetlands, including vast tule marshes that could be inundated for several contiguous months during periodic high flow events (Kelley 1989). In 1850, conversion of the lower Sacramento River and the Delta to agricultural uses was stimulated by passage of an act that extended the Swamp Lands Act of 1849 to California and 11 other states. Under this legislation states were granted land rights to swamps and overflow areas if they reclaimed them and thus increased the potential productivity of what was perceived in that era as wasted and unproductive land.

A century and a half of land use conversion caused the direct loss and fragmentation of riparian, wetland, and off-channel habitat. It also likely increased fine sediment loading in the mainstem Sacramento River by increasing soil erosion from exposed fields. The disturbance associated with land use conversion, coupled with the import of non-native vegetation, also provided a pathway for alien species to establish in the valley.

2.1.2 Water impoundment and diversion

Prior to the development of large-scale water supply dams in the mid-twentieth century, miners and settlers constructed smaller dams to impound and divert water for mining, irrigation, and grazing in the mid- and late nineteenth century. Though these low-head dams did not significantly alter the hydrologic or sedimentologic regime of the Sacramento River, they did block the upstream passage of adult salmonids so that these fishes could not access their historical spawning grounds. The California Fish Commission documented an early example of a seasonal passage barrier in the upper Sacramento River, citing a mining diversion tunnel located upstream of the confluence with the Pit River that impeded upstream access of fall-run Chinook salmon (*Oncorhynchus tshawytscha*) during low-flow periods in August and September (CFC 1890, as cited in Yoshiyama et al. 1996). More permanent passage barriers were constructed by the Pacific Gas and Electric Company (PG&E) in the 1920s as part of hydroelectric facilities in the Pit River (Vestra 2004). Though these upper watershed dams blocked access to historical spawning grounds, there was still significant spawning habitat available downstream until 1917, when the construction of the seasonal ACID Dam at Redding began impeding access to the majority of historical spawning habitat in the Sacramento River. Typically operated between April and October to provide irrigation water, the ACID Dam blocked the upstream migration of spring-run Chinook salmon, and a portion of the winter-run and fall-run Chinook salmon migration.

2.1.3 Dam construction

The construction of Shasta Dam began in 1938[1] as part of the Central Valley Project (CVP), inaugurating an era of large-scale water supply and flood control dams in California and the

---

[1] Different dates are given for the beginning of construction. Some sources say Shasta Dam construction began in 1940, which is when contractors began pouring concrete to form the dam. Excavation of the dam site and work on the abutments began in 1938, and these activities likely had an effect on the river (e.g., by increasing turbidity).

beginning of the most pronounced changes in the Sacramento River corridor. The dam was completed and began to impound water in 1945. During the construction phase, flow was temporarily diverted around the construction site by coffer dams, which produced no significant effect on flow magnitudes and only a small delay in the timing of flow peaks. In contrast, the effects of Shasta Dam construction on the sediment regime of the Sacramento River were substantial and almost immediate. More than 7.0 million yd^3 (5.35 million m^3) of gravel and sand were mined from the upper Sacramento River basin to support construction of the dam and related infrastructure, which reduced the volume of coarse sediment stored in the channel and supplied by tributaries. Figure 2-2 illustrates the scale and the lasting effects of aggregate mining conducted along the Sacramento River to support dam construction, showing the large mining pit at Kutras Park, a primary borrow area. Also significant were the effects of dam-related blockage of sediment from the upper watershed. This began as early as 1940, when temporary cofferdams were constructed to divert water around the construction site. These cofferdams probably interrupted sediment transport from the upper watershed to the reach below the dam construction site during the high flow events of 1940 (186,000 cfs), 1941 (82,300 cfs), and 1942 (85,000 cfs) (K. Buer, pers. comm., 2005).

Keswick Dam construction began in 1941. Although it was not completed until 1950, by 1942 it defined the upstream limit of anadromy in the Sacramento River. Keswick Dam was equipped with a fish trap in 1943 to facilitate the harvest of salmon for artificial propagation at Coleman National Fish Hatchery on Battle Creek, as part of a mitigation plan for Shasta Dam.

Whiskeytown Dam was completed on Clear Creek in 1963, as part of the Trinity River Division of the CVP. Whiskeytown Dam impounds water imported via inter-basin transfer from the Trinity River. Except for minimum and flood flow releases into the Clear Creek channel, water stored in Whiskeytown Lake is routed to Keswick Dam via the Spring Creek Tunnel to generate electricity. Beginning in 1964, an average of 11.2 million yd^3 (8.6 billion m^3) of cool Trinity River water was diverted into the Sacramento River, thereby increasing flow volumes and decreasing water temperatures in the mainstem channel (USDA Forest Service 2005).

Once these large water supply and re-regulation dams were completed, they not only blocked access to more than 80% of historical salmonid spawning habitat in the upper Sacramento River (Lindley et al. 2006), but they also trapped sediment from the upper watershed. Other dam-related modifications to the system included dramatic changes in the flow and water temperature regimes and changes in large woody debris (LWD) loading of the Sacramento River.

2.1.4 Aggregate mining

In addition to the greater than 7 million yd^3 (5.35 million m^3) of sediment mined from the upper Sacramento River basin to support the construction of Shasta and Keswick dams in the 1940s, continued aggregate mining removed several million more cubic yards of sand and gravel from the Sacramento River and its tributaries to support urbanization in the north Sacramento Valley and the construction of Interstate 5 (CDWR 1980). CDWR estimates that between 1 and 1.5 million yd^3 (0.8 and 1.1 million m^3) of locally mined aggregate were used in the 1960s to construct the reach of Interstate-5 between Red Bluff and Corning (Buer 1984). A review of mining records allowed CDWR to estimate that, on average, 1.8 million yd^3 (1.4 million m^3) of aggregate are mined annually from Shasta and Tehama counties, primarily from tributaries of the Sacramento River (CDWR 1980, Buer 1984, Buer 1994a), to support general urban needs.

Aggregate mining decreases the volume of sediment stored in the channel and floodplain of the Sacramento River, and decreases the coarse sediment load from tributaries, thereby depriving the mainstem channel of a fundamental building block of aquatic habitat. Remnant mining pits can also serve as sediment traps that disrupt the routing of bedload, and may harbor non-native salmonids predators (e.g., large-mouth bass).

2.1.5 Flood control levees and bank armoring

The construction of Shasta and Whiskeytown dams improved flood management in the northern Central Valley, which in turn encouraged further development of floodplain lands that previously had been vulnerable to periodic flooding. Beginning approximately 160 years ago with the discovery of gold, early settlers began to construct what ultimately became an extensive network of levees throughout the Central Valley to control flooding and reclaim land for agriculture. By 1893, mining sediment from hydraulic mining had significantly reduced the natural flood conveyance capacity of the mainstem river to the extent that valley floor farming was threatened. The California Debris Commission was created to address the issue. In 1911 the commission was transformed into the State Reclamation Board, which was charged with regulating the network of private levees which had been constructed in a piecemeal fashion throughout the valley.

The Sacramento River Flood Control (SRFC) Project was authorized by Congress in 1917 to formalize a more coordinated flood control effort. Construction of SRFC Project levees by the Army Corps of Engineers was not completed until 1965 (USACE 1999). Project levees were constructed to alleviate flooding issues and also increase the river's sediment transport capacity. The increased sediment transport capacity was desired as a means to pass gold mining debris down through (and out of) the system. Project levees worked so well in achieving this goal that, in 1960, Congress authorized the Sacramento River Bank Protection Project to protect the Project levees from exacerbated bank erosion (USACE 1999). Continuing agricultural land and infrastructure development (including the building of roads and diversion facilities) also increased the need for bank erosion control. The first significant effort to prevent bank erosion in the valley began in 1963 as part of the U.S. Army Corps of Engineers' (USACE) Chico Landing to Red Bluff Bank Protection project, which placed revetment on more than 70,000 linear ft (20,000 linear m) of bank between RM 243 and RM 194 through 1985 (CDFG 1983).

By preventing bank erosion, revetment reduces the recruitment of gravel stored in channel banks, (Buer 1994b). Bank armoring can induce channel narrowing and incision, which can reduce spawning and rearing habitat of salmonids (USFWS 2000). The large boulders used to stabilize a bank may also confer a competitive advantage to piscivorous fish, especially ambush predators that lurk in the interstices. Large riprap may provide cover habitat for salmonids in some areas (Lister et al. 1995), but water velocity in proximity to large riprap may offset any benefits and partially explain the low use found in some studies (Ecos 1991). Bank armoring also prevents progressive channel migration, which can reduce the formation of off-channel habitats by reducing the susceptibility of meander bends to cutoff. A further impact of revetment is that it prevents the formation of fresh vertical cutbanks, which bank swallows (*Riparia riparia*) require for nesting.

2.1.6 Red Bluff Diversion Dam

Red Bluff Diversion Dam was completed in 1964 at RM 243.5 to divert water into the Tehama-Colusa and the Corning canals, which provide irrigation water to farmers in the Sacramento Valley. Red Bluff Diversion Dam is equipped with a series of gates that are seasonally lowered to impound water, and that lead to the creation of Lake Red Bluff. The increase in stage caused by the dam allows water to flow into the canals. When the irrigation season is over, the RBDD gates can be raised to allow water and sediment to pass downstream and to allow fish to move upstream.

Red Bluff Diversion Dam was equipped with a fish passage structure that permitted counts to be made of the number of adult salmon returning to spawn in the upper river each year. These counts have been used to develop population estimates for the different runs of salmonids. It was noted fairly early that even with the fish passage structure in full operation, the RBDD can impede upstream access when the gates are lowered (Hallock and Fisher 1985). In an effort to mitigate its effects on listed fish species, RBDD gate operations have been adapted over time. For example, in 1987 the U.S. Bureau of Reclamation (USBR) began raising the RBDD gates between December and April to provide easier upstream passage for winter-run Chinook salmon, which were being evaluated as a candidate species for federal listing. As a result of the change in gate operations, the fish passage structure no longer provides for continuous tracking of salmonid escapements. In 1990, a fish ladder was constructed in the middle of RBDD so that fish could access upstream reaches when the gates are lowered for the irrigation season.

To help mitigate the loss of spawning habitat in the upper Sacramento River, the USBR placed gravel in the first 3.2 mi (5.1 km) of the Tehama-Colusa Canal in hopes that this area could function as an artificial spawning site for Chinook salmon beginning in 1971 (USRSSAC 1983). However, few salmon used the canal for spawning, and some of the associated infrastructure posed a hazard for juvenile salmonids migrating downstream, so the spawning channel was abandoned.

The fish ladder and subsurface openings beneath the RBDD gates allowed juvenile salmonids to pass downstream. However, state and federal fisheries agencies expressed concern that juveniles were becoming disoriented as they passed through the turbulent flows of the fish ladder and the narrow openings beneath the dam gates. Such disorientation would presumably make them easy prey for congregations of Sacramento pikeminnow (*Ptychocheilus grandis)* downstream of RBDD (Hallock and Hall 1977, Vogel et al. 1988). The lentic conditions of Lake Red Bluff when dam gates are down may also provide habitat conditions that favor piscivorous predators rather than juvenile salmonid emigrants. Currently, RBDD gates are raised between mid-September and mid-May to facilitate both the upstream passage of adult salmonids and the downstream migration of juveniles; however, there continues to be concern about impacts of this structure on a suite of migratory fishes (e.g., green sturgeon, lampreys) in addition to salmonids.

2.2 Effects on Watershed Inputs and Ecosystem Processes

The human activities described in the previous section altered the flow regime and the routing of material (e.g., gravel, water, and large woody debris) in the Sacramento River, with concomitant effects on habitats and habitat conditions. Alluvial rivers are dynamic systems that are affected by complex interactions between numerous inputs and processes. A simplified conceptual model illustrating these interactions is shown in Figure 2-3.

In the model, natural watershed inputs (such as water, sediment, and nutrients) drive physical processes (such as sediment transport and channel migration) that, in turn, determine geomorphic attributes and physical habitat structure of the river-floodplain system. The geomorphic attributes and habitat structure drive biological responses and are important determinants of plant and animal species abundance, distribution, and composition. Modification of any of the key inputs or processes will influence channel and floodplain geomorphic attributes and, subsequently, affect plant communities and fish and wildlife populations. For example, reduction in peak flows (a watershed input) can alter fluvial processes such as the timing, frequency, extent, and duration of floodplain inundation. This alteration in inundation patterns can result in changes in riparian plant species composition and age-class structure, which in turn can alter habitat suitability for native birds and thus result in a shift in bird community species composition. In turn, riparian vegetation can feed back to hydraulic and geomorphic processes. For example, increased roughness provided by newly established vegetation can increase sediment deposition and floodplain accretion, and encroachment of vegetation into the active channel following flow regulation commonly contributes to channel deepening.

2.2.1 Changes in watershed inputs

This section provides a general description of the effects of human disturbances on key watershed inputs depicted at the top of Figure 2-3. More detailed discussions of human induced changes to watershed inputs are included in Chapter 3 and the focal species Chapters (4–9).

2.2.1.1 Water and energy

In rivers draining the northern Central Valley, natural flow conditions are characterized by low flows in summer and early fall, large but brief flow peaks in winter caused by rain storms and rain-on-snow events, and a modest spring snowmelt. Each component of the natural hydrograph drives processes that shape and sustain the river-floodplain system. Alteration of any of these components can potentially alter the river ecosystem structure and function.

River flows are both a key watershed input and a key source of energy in the Sacramento River. Flows transport other key watershed inputs (e.g., sediment, LWD, and seeds) to create and maintain aquatic, floodplain, and riparian habitats. Flows are also a key determinant of habitat conditions.

The water resources of the Sacramento River basin have been the focus of intense human development for the past century. The North-of-the-Delta Offstream Storage Technical Advisory Group (NODOS TAG) delineated five distinct periods of hydrologic alteration in the Sacramento River (NODOS TAG 2004):

- **1892 to 1938**. This period defines the pre-dam era, beginning with the availability of historical discharge data at the Bend Bridge gauge (U.S. Geological Society [USGS] # 11377100) and ending with the initiation of Shasta Dam construction.
- **1939 to 1944**. This period defines the time when Shasta Dam was being constructed.
- **1945 to 1964**. This period defines the initial operation of the CVP, which included the Shasta and Delta divisions of the project. The Shasta Division consisted primarily of Shasta and Keswick dams. The Delta Division included diversion and conveyance facilities in the Delta, such as the Tracy Pumping Plant, the Contra Costa Canal, the Delta Cross Channel, and the Delta-Mendota Canal. During this period, flow releases from Shasta Dam were routed to the south Delta where water was pumped into the canals to support agriculture in the San Joaquin Valley and Contra Costa County.
- **1965 to 1992**. This period defines the expansion of CVP facilities and water deliveries as the Sacramento River and Trinity River divisions became operable. The primary components of the Sacramento River Division were the Red Bluff Diversion Dam, the Tehama-Colusa Canal, the Glenn Colusa Canal (operated by the Glenn Colusa Irrigation

District, or GCID), and the Corning Canal and Pumping Plant. The primary components of the Trinity River Division were Trinity Dam, Lewistown Dam, the Clear Creek Tunnel and Judge Francis Carr Power plant, Whiskeytown Dam, and the Spring Creek Tunnel and Powerhouse. The Trinity Division began importing an average of 1.2 MAF into the Sacramento River during this period. This period also marks the beginning of State Water Project (SWP) operations (e.g., Oroville Dam on the Feather River and the California Aqueduct) and joint operations between the SWP and CVP via the San Luis Reservoir, both of which fall outside of the study area for this SOS Report but nevertheless influenced flow releases to the Sacramento River.

- **1993 to 1998**. This period defined changes to CVP operations mandated by US Fish and Wildlife Service (USFWS) protections for winter-run Chinook salmon and Delta Smelt, and water quality requirements in the Delta. Diversions from the Trinity Division were also reduced during this period to support flow studies and restoration in the Trinity River.

Though diversion facilities constructed in the early twentieth century (e.g., ACID Dam, Glen Colusa Irrigation District [GCID] diversion) altered natural flow patterns in the Sacramento River, the completion and operation of Shasta Dam stimulated the most significant changes in the flow regime beginning in 1945 (Kondolf et al. 2000). The general effect of the dam was to reduce the magnitude and frequency of high flow events in the winter and spring and to increase summer and fall base flows, which reflects the purpose of the dam to impound water for release during the irrigation season and to increase flood protection during the rainy season. Figure 2-4 illustrates the changes in seasonal flow patterns caused Shasta Dam operations by comparing mean monthly discharge at the USGS gauge at Bend Bridge (# 11377100) before and after construction of the dam. Figures 2-5 through 2-7 provide additional illustrations of how the operation of Shasta Dam has altered the natural flow regime in the Sacramento River.

By reducing the magnitude and frequency of winter storm events, flow regulation has reduced the energy available to drive several ecological processes in the river-floodplain system. By increasing the magnitude of summer baseflows, dam operations have changed the shape of the hydrograph from a gradual recession limb to an artificially elevated plateau. Such changes in the pattern of flows can affect the establishment, distribution, composition, and survival of naturally recruited riparian vegetation. For example, the spring recessional period has been curtailed by water management operations that rapidly decrease river stage during a period when riparian vegetation is attempting to colonize point bars. This leads to establishment of seedlings at much lower elevations on point bars than current ecological models suggest are appropriate for colonization. Such seedlings are likely exposed to repeated inundation which may result in high mortality. In addition, an elevated water table caused by artificially high base flows may prevent recently recruited riparian vegetation from growing root structures deep enough to tap the water table when irrigation deliveries are reduced abruptly, thereby inducing mortality. These ecological interactions are discussed in greater depth in Chapter 9.

2.2.1.2 Coarse sediment supply

Sediment is a fundamental building block of river systems, providing material for construction of riffles, bars, banks, and floodplains. In large undisturbed watersheds, sediment is supplied from upstream sources (such as slopes and tributaries) and after being temporarily stored in alluvial reaches, is transported to a depositional zone in a downstream delta. If sediment supply is roughly equal to the river's sediment transport capacity, a condition of “dynamic equilibrium” will develop (Schumm 1977). Under dynamic equilibrium individual reaches of the river may change dramatically over time, even though sediment erosion and deposition are balanced for the river as a whole over the long term.

Shasta Dam traps all of the coarse material that is supplied from upstream sediment sources. The estimated annual average load of coarse sediment of those sources is 50,000 yd^3 (38,000 m^3) (K. Buer, pers. comm., 2005). This implies that the reach below Keswick Dam has been deprived of an estimated 3 million yd^3 (2.3 million m^3) since Shasta Dam construction in 1945. The deficit of coarse sediment from the upper watersheds was exacerbated by the nearly 7 million yd^3 (5.35 million m^3) of sediment that was mined from the river and floodplains for dam building (CDWR 1980), and the 1.8 million yd^3 (1.4 million m^3) of aggregate that was mined to support urbanization of Redding (CDWR 1980, Buer 1984).

The first significant source of coarse sediment below Keswick Dam (RM 302) is Cottonwood Creek (RM 273.5). Tributaries between Keswick Dam and Cottonwood Creek contribute relatively little sediment to the mainstem channel, because they drain small basins composed of resistant material or are themselves regulated by dams and have been mined for aggregate (e.g., as is the case with Clear Creek) (CDWR 1980). Erosion of banks can often supply rivers with coarse sediment, but along the upper Sacramento River, much of the channel is bounded by bedrock or other erosion-resistant material (CDWR 1980), such that supply of material from bank erosion is insufficient to offset the overall deficit of coarse sediment.

2.2.1.3 Large woody debris

Large woody debris (LWD) deposits in river channels contribute to aquatic habitat complexity. In healthy, meandering river systems, erosion of channel banks and floodplains on the outside of meander bends, undercuts mature riparian vegetation such that it falls into the channel. Flow velocities are generally higher on the outside of bends, such that the rearing habitat potential is low for fish like juvenile salmonids in the absence of complex bank habitat (e.g., LWD). Juvenile salmonids prefer zones where slow velocity water is immediately adjacent to high velocity water. The low velocities enable minimal energy expenditure as they maintain position, whereas the higher velocities deliver insect drift which serves as an important food source. Because LWD is often recruited on the outside of meander bends (where bank erosion and undercutting of riparian vegetation is greatest), it can create velocity refugia within the high velocity core, providing ideal conditions for juvenile salmonids, and effectively increasing rearing habitat in the channel.

Rates of LWD recruitment in the Sacramento River are probably much lower than they were historically (Henderson 2003). The conversion of riparian forest to agriculture on the one hand works to accelerate bank erosion and cut off processes (see Chapter 3). This would generally be expected to increase LWD recruitment, but the clearing of the forests has also reduced the amount of mature woody vegetation on banks, such that bank erosion probably yields less LWD on average. Bank armoring activities have probably reduced LWD recruitment by locally reducing bank erosion rates (USFWS 2000). The reduced magnitude and frequency of winter high flow events would generally work to decrease bank erosion rates along unprotected banks, but as discussed in Chapter 3, there are many factors that may have affected bank erosion on the Sacramento River and the net effect on LWD recruitment is difficult to precisely quantify.

2.2.2 Changes in fluvial geomorphic processes

In this section we present a brief overview of geomorphic processes, highlighting important changes that have occurred in the evolution of the Sacramento River and his floodplains over the last 150 years. We elaborate on these recent changes in Chapter 3, where geomorphic processes are discussed in depth, and also in the focal species chapters (Chapters 4–9), where the evolution of individual habitat types are discussed.

2.2.2.1 Sediment transport, deposition, and scour

Sediment transport, deposition, and scour regulate the formation of key habitat features such as point bars and gravel deposits. These processes are regulated by the magnitude and frequency of flow, with relatively large floods providing the energy required to mobilize sediment from the bed. The threshold flow for mobilization of sediment depends on channel morphology (i.e., width, depth, and slope) and the grain-size distribution of the sediment. On the Sacramento River below Keswick Dam (RM 302), the threshold flow for mobilization of spawning gravel has been estimated to be about 50,000 cfs, based on observations of mobilization of injected gravel during floods stages (see Chapter 3).

In undisturbed alluvial rivers, channels and bedforms evolve in response to flow and sediment loading conditions that may vary from moment to moment by orders of magnitude. In many cases, the frequency distribution of flow and sediment supply are such that rivers convey the greatest fraction of their load at an intermediate "dominant" discharge, which is often close to the bankfull flow (Wolman and Miller 1960, Leopold et al. 1964). Though the recurrence interval of bankfull flow varies from river to river, it is often close to 1.5 to 2 years (Leopold et al. 1964). This provides a rational basis for assuming that coarse sediment is routed as bedload during the 1.5-year flood. Flow regulation of the Sacramento River has reduced its $Q_{1.5}$ by 30% from 86,000 cfs (2,400 $m^3 s^{-1}$) to 61,000 cfs (1,700 $m^3 s^{-1}$) (Kondolf et al. 2000).

Whereas bankfull flow may provide a good first approximation for assessing the threshold for bed mobilization, it is not necessarily indicative of flows that are required to maintain the health of habitats in the alluvial system. For example, it has been estimated that the naturally occurring 5- to 10-year recurrence interval flood may often be required for maintenance of a mobile alternating bar-pool sequence (Trush et al. 2000)—a desired condition from an ecological standpoint. In the regulated flow regime on the Sacramento River, the 10-year flood has been reduced by 38% from 218,000 cfs (6,170 $m^3 s^{-1}$) to 134,000 cfs (3,790 $m^3 s^{-1}$) (Kondolf et al. 2000).

At many locations between Keswick Dam and Red Bluff (RM 302–243) (Figure 1-2), the channel is characterized by bedrock control of its baselevel and of its banks. This implies that, compared to alluvial reaches downstream, the channel between RM 302 and RM 243 has been limited in its ability to adjust hydraulic geometry (i.e., channel width and depth) in response to dam-related changes in flow. Ultimately, it is difficult to rule out the possibility that the channel is not in balance with the current flow regime, such that typical recurrence intervals of mobilization and bed form alteration are much longer than they were before the dams reduced magnitude of $Q_{1.5}$ and Q_{10}. This implies that the bed and point bars may have become static in the post-dam era, such that fossilized remnants of gravel are all that remains of once abundant spawning habitat in winter-run spawning reach (see Chapters 3–4 for further details).

The flow required for mobilization and scour a channel bed depends in part on the grain-size distribution of the bed sediment. On the Sacramento River the grain-size distributions of deposits between Keswick Dam (RM 302) and Cottonwood Creek (RM 273.5) may have increased since the construction of Shasta Dam, due to winnowing associated with dam-related reductions in sediment supply (see Chapter 3 for further details). This would tend to increase the threshold for mobilization and scour of the channel bed, even as the frequency of high flows was reduced by the operations of Shasta Dam. The hypothesized coarsening of the bed would thus tend to make mobilization of sediment and bedforms even less likely under the regulated flow regime in the upper Sacramento River.

Relative to the upper Sacramento River (RM 302–343) there is significantly less bedrock control between Red Bluff (RM 243) and Colusa (RM 143), though a significant portion of this river reach is constrained by bank armoring. Because the channel between RM 243 and RM 143 is largely alluvial, it may be capable of adjusting its dimensions and lateral position in response to changes in flow conditions. This is supported by widespread evidence of frequent lateral migration in the middle Sacramento River (e.g., Micheli et al. 2004). This implies that the middle Sacramento River experiences much more frequent bed and bar mobilization than the upper Sacramento River.

2.2.2.2 Channel migration and bank erosion

Progressive meander migration and meander bend cutoffs in the middle Sacramento River are driven by flow, which provides the energy to erode banks and scour new channels across floodplain surfaces. As discussed in greater detail in Chapter 3, the effects of human activities on meander migration rates of the Sacramento River are complicated. As a general rule, the reduced magnitude and frequency of winter floods, in addition to bank armoring, should work to reduce rates of bank erosion. However, because low flows can contribute to incremental bank erosion, the increase in summer base flows in the post-dam era is an important confounding factor to consider (see Chapter 3). Moreover the effects of changes in flow on meander migration rates are not always clear because bank erosion is influenced by many factors including soil cohesion and vegetation (e.g., Buer 1994a, Micheli et al. 2004). There are now several strong indications, for example, that meander migration rates have been accelerated by agricultural conversion of riparian habitat to open fields and orchards. This is presumably due to an increase in erodibility associated with loss of effective cohesion and roughness on the floodplains. Changes in erodibility of banks have also occurred in the post-dam era. Bank armoring activities, which are designed to stop bank erosion, have been constructed along many stretches of the actively migrating middle Sacramento River. For example, rock revetment has been placed on more than 70,000 linear ft (20,000 linear m) of bank in the reach between Red Bluff and Chico Landing (RM 243–193). As we will show in Chapter 3, the competing effects of bank protection, vegetation clearing, and flow reductions in the post-dam era are difficult to disentangle from one another. Overall these effects appear to have produced a slight overall increase in meander migration rates and a more pronounced increase in the relative importance of meander bend cutoffs versus progressive meander migration, which are discussed in more detail in Chapter 3.

2.2.3 Hydrology of the floodplain

Inundation of floodplains reduces flood flow magnitude and promotes exchange of nutrients, organisms, sediment, and energy between the terrestrial and aquatic systems. Flood pulses contribute to high rates of primary productivity in functioning floodplain systems (Junk et al. 1989). On the Sacramento River, floodplains provide important winter and spring spawning and rearing habitats for native fish, such as Sacramento splittail (*Pogonichthys macrolepidotus*) and Chinook salmon (*Oncorhynchus tshawytscha*) (Moyle et al. 2004, Sommer et al. 2001).

Typically, the floodplain immediately adjacent to the river is maintained at an elevation equal to the bankfull stage of the channel, such that discharge magnitudes greater than the bankfull flow inundate the adjacent floodplains (Wolman and Leopold 1957, Leopold et al. 1964). As bankfull flow typically has a recurrence interval of 1.5–2 years on the alluvial rivers, flow magnitudes greater than the $Q_{1.5}$ flow event are often assumed to initiate floodplain inundation.

Most of the upper Sacramento River between Keswick Dam (RM 302) and Red Bluff (RM 243) is bounded by high banks and terraces, such that there is little opportunity for floodplain inundation in this reach. Over much of the lower portion of the study reach, between Chico Landing (RM 193) and Colusa (RM 143) the river is bounded by levees which provide flood protection for cities and agricultural areas. However, the levees of the middle Sacramento River are for the most part setback from the mainstem channel such that flooding can be significant within the river corridor. In the middle Sacramento River between Red Bluff (RM 243) and Chico Landing (RM 193), the mainstem channel is flanked by broad floodplains. Evidence of ongoing sediment deposition of these areas testifies to continued inundation of floodplains in this reach (Buer 1994b).

Reductions in the frequency and magnitude of winter floods have presumably reduced the extent and frequency of floodplain inundation throughout the Sacramento River study area. However this is difficult to verify in the absence of quantitative data on floodplain inundation for the pre-dam era. Even if that kind of data was available, the confounding effects of differences in local conditions (due to effects of levees and riprap, which were constructed progressively over the last century) would be difficult to disentangle from the effects of changes in the frequency and magnitude of winter floods. Additional considerations regarding overbank flow and its effects on hydrogeomorphic processes in the pre- and post-dam and eras are provided in Chapter 3.

To the extent that reductions in winter flow and levees have reduced floodplain inundation in the middle Sacramento River, rates of groundwater recharge on the floodplain have probably also been reduced. This would tend to reduce the average elevation of the groundwater table over time. Quantitative confirmation of this is difficult in the absence of historical data. Also difficult to confirm is the expectation that the groundwater table in the immediate vicinity of the channel has risen due to the increased magnitude of baseflows in summer and fall in the regulated flow regime. If the groundwater table has indeed risen in the immediate vicinity of the channel, it might have important implications for establishment of naturally recruited riparian vegetation (see Chapter 9 for further discussion).

2.3 Restoration Activities

In the past two decades, human activities have accelerated to conserve and restore the resources of the Sacramento River. A variety of approaches have been used. In many cases, activities have emphasized an ecosystem-based approach in which watershed inputs (e.g., gravel and flow) are restored such that habitats can be created and maintained by natural riverine processes. Others have focused on planting of native species, as a means to provide natural habitat for species of concern and strive toward outpacing the spread of non-native invasive species. The traditional engineering approach in which waterways are physically reconstructed by humans is also sometimes used. For example, there is increasing support from ecologists and managers alike for the implementation of levee setback strategies, in which flood control structures are repositioned at a greater distance from the main channel. Some of the key restoration activities of the Sacramento River are described briefly below.

2.3.1 Habitat conservation and restoration

Several organizations have purchased thousands of acres of land in the Sacramento River corridor for habitat conservation and restoration, aligning their conservation efforts with the goals and objectives of the Sacramento River Conservation Area Forum (SRCAF). For example, the Sacramento National Wildlife Refuge Complex (the "Refuge") has protected over 65,000 ac (26,000 ha) of riparian, upland, and wetland habitat in the northern Central Valley by obtaining fee title and conservation easements. Meanwhile, agencies and non-profit organizations like The Nature Conservancy have purchased approximately 15,000 ac (6,000 ha) along the Sacramento River in the past decade. In the process, nearly 3,600 ac (1,500 ha) have been restored to native riparian forest (Petersen et al. 2003, Alpert et al. 1999, R. Luster, pers. comm. 2006)

Land acquisition for conservation and restoration are a significant step toward attaining the shared vision of promoting a healthy, contiguous riparian zone bordering a meandering Sacramento River between Red Bluff (RM 243) and Colusa (RM 143) (SRCAF 2003). The conservation and restoration of native riparian vegetation has been successful in providing important foraging and breeding habitat for special-status species including birds (e.g., western yellow-billed cuckoo [*Coccyzus americanus occidentalis*], Swainson's hawk [*Buteo swainsoni*], bank swallow), mammals (e.g., western mastiff bat [*Eumops perotis*]), pallid bat [*Antozous pallidus*]) and insects (e.g., valley elderberry longhorn beetle [*Desmocerus californicus dimorphus*]) (Small et al. 2000, TNC 2005).

2.3.2 Gravel augmentation

Concern about dramatic declines in Chinook salmon populations in the late 1970s and early 1980s stimulated a series of gravel augmentation projects on the upper Sacramento River. The aim of the added gravel was to counteract the effects of reduced sediment supply caused by dam construction and aggregate mining. Since 1978, approximately 242,000 yd^3 (185,000 m^3) of spawning-sized gravel has been added to the channel of the Sacramento River between Keswick Dam (RM 302) and the confluence with Clear Creek (RM 289.2).

As discussed in greater detail in Chapter 3, it is difficult to know if the recent gravel augmentation has had an appreciable effect on in-channel gravel storage. The added gravel amounts to a small fraction of the > 10 million yd^3 (7.6 million m^3) of sediment that was mined from the basin and trapped by the dams. However there is some indication, as discussed in Chapter 3, that the gravel injections may have helped maintain salmonid spawning habitat locally, within the immediate vicinity downstream of the injection sites.

2.3.3 Water temperature management

The severe drought of 1976 and 1977 exhausted the reservoir of cold water in Shasta Lake, such that warm water releases from the dam created water temperature conditions in the Sacramento River that contributed to declines in several runs of Chinook salmon (Hallock and Fisher 1985). The effects of these warm water releases underscored the difficulty of maintaining cold pool storage for subsequent years while at the same time providing cold water releases to protect incubating salmonid eggs.

Since 1993, Shasta Dam operations by USBR have attempted to provide water temperatures of 56°F (13°C) in the Sacramento River between April 15 and September 30 to protect incubating eggs of the endangered winter-run Chinook salmon. To protect emergent fry and juveniles in the month of October, the target is slightly higher at 60°F (16°C). Competing demands of water have often prevented maintenance the targeted water temperatures at Bend Bridge (RM 260.5), the current compliance point for the USBR operations. The Shasta Temperature Control Device (TCD), completed in 1997, allowed USBR release of cold water from the reservoir through penstocks and thus minimize disruption of hydroelectric production while providing cold water for the river. Though we did not have the tools to evaluate the success of the TCD in achieving water temperature targets, the USBR water temperature model could be applied for such an assessment.

2.3.4 Other restoration activities

A diverse array of additional restoration activities have been conducted along the Sacramento River corridor. For example, TNC and the USFWS have worked with the USFWS Sacramento River National Wildlife Refuge to reconstruct distributary channels at RM 217 (in the Rio Vista Unit), RM 194 (in the Capay Unit), and RM 199 (in the Pine Creek Unit). The USFWS has also removed a private levee on Refuge property at RM 217 (in the Rio Vista Unit), and retired private bank armoring at RM 232.5 (in the Flynn Unit). This retirement of bank armoring promoted erosion of a new cutbank and corresponding deposition of a gravel bar. Biological responses to the cutbank and deposition included renewed activity by spawning salmonids and establishment of the second largest nesting bank swallow colony ever documented on the river.

Restoration via levee setback is also being considered as a management alternative (Larsen et al. 2006, Golet et al. 2006). This approach has the advantage of retaining or even enhancing flood control benefits of levees while at the same time restoring habitat. For example, TNC has been collaborating since 2000 with several federal, state, and local partners to develop an ecosystem restoration and flood damage reduction project near flood-prone Hamilton City and the surrounding agricultural lands. A final feasibility study describes the project in detail (USACE 2004). As part of the plan, a 6.7 mile long stretch of levee will be set back, such that 1,500 ac (600 ha) can be revegetated and hydrologically reconnected to the Sacramento River. In the cost-benefit analysis of the plan, calculations include not only the economic benefits reaped by enhanced flood protection, but also assign value to the ecological benefits of revegetation and hydrologic connectivity. As a result the cost-benefit ratio is about 1:1.8. Project construction is scheduled to begin in 2008.

2.4 References

Alpert, P., F. T. Griggs, and D. R. Peterson. 1999. Riparian forest restoration along large rivers: initial results from the Sacramento River Project. Restoration Ecology 7: 360–368.

Bay Institute, The. 1998. From the Sierra to the sea: the ecological history of the San Francisco Bay-Delta watershed. San Rafael, California.

Buer, K. 1984. Middle Sacramento River spawning gravel study. California Department of Water Resources, Northern District, Red Bluff.

Buer, K. 1994a. Use of alternative gravel sources for fishery restoration and riparian habitat enhancement in Shasta and Tehema counties, CA. Internal memorandum to J. Siperek, G. Stacey, and T. Mills from K. Buer, Chief, Geology Section, California Department of Water Resources, Northern District, Red Bluff. August 1.

Buer, K. 1994b. Sacramento River bank erosion investigation memorandum progress report. Internal memorandum to R. Scott and L. Brown from K. Buer, Chief, Geology Section, California Department of Water Resources, Northern District, Red Bluff.

CDFG (California Department of Fish and Game). 1983. Sacramento River and tributaries bank protection and erosion control investigation-Evaluation of impacts on fisheries. The Resources Agency, Department of Fish and Game, Bay-Delta Fishery Project.

CFC (California State Board of Fish Commissioners). 1890. (11th) Biennial report for 1888–1890. CFC, Sacramento.

CDWR (California Department of Water Resources). 1980. Upper Sacramento River spawning gravel study. Report. Prepared for California Department of Fish and Game by CDWR, Northern District, Red Bluff.

Ecos, Inc. 1991. Biological data report regarding Sacramento River bank protection project impacts on winter-run Chinook salmon. DACW05-88-D-0058. Prepared for U.S. Army Corps of Engineers, Sacramento, California.

Golet G. H., M. D. Roberts, E. W. Larsen, R. A. Luster, R. Unger, G. Werner, and G. G. White. 2006. Assessing societal impacts when planning restoration on large alluvial rivers: A case study of the Sacramento River Project, California. Environmental Management 37:862-879.

Henderson, A. W. 2003. Tracking large woody debris in the Sacramento River: An application of radio telemetry. Master's thesis. California State University, Chico.

Hallock, R. J., and F. Hall. 1977. A discussion of Sacramento squawfish predation problems at Red Bluff Diversion Dam. Memorandum. California Department of Fish and Game, Anadromous Fisheries Branch, Red Bluff.

Hallock, R. J., and F. W. Fisher. 1985. Status of the winter-run Chinook salmon, *Oncorhynchus tshawytscha*, in the Sacramento River. Anadromous Fisheries Branch Office Report. California Department of Fish and Game.

Junk, W. J., P. B. Bayley, and R. E. Sparks. 1989. The flood pulse concept in river-floodplain systems. Pages 110–127 in D. P. Dodge, editor. Proceedings of the international large river symposium. Canadian Special Publication of Fisheries and Aquatic Sciences 106.

Kondolf, G. M., T. Griggs, E. W.Larsen, S. McBain, M.Tompkins, J. G. Williams, and J. Vick. 2000. Flow regime requirements for habitat restoration along the Sacramento River between Colusa and Red Bluff. CALFED Bay Delta Program, Integrated Storage Investigation, Sacramento, California.

Kelley, R. L. 1989. Battling the inland sea: floods, public policy, and the Sacramento Valley, 1985-1986. University of California Press, Berkeley.

Larsen, E. W., E. H. Girvetz, and A. K. Fremier. 2006. Assessing the effects of alternative setback channel constraint scenarios employing a river meander migration model. Environmental Management 37: 880–897.

Leopold, L. B., M. G. Wolman, and J. P. Miller. 1964. Fluvial processes in geomorphology. W. H. Freeman and Company, San Francisco, California.

Lindley, S. T., R. S. Schick, A. Agrawal, M. Goslin, T. E. Pearson, E. More, J. J. Anderson, B. May, S. Greene, C. Hanson, A. Low, D. McEwan, R. B. MacFarlane, C. Swanson, and J. G. Williams. 2006. Historical population structure of Central Valley steelhead and its alteration by dams. San Francisco Estuary and Watershed Science 4: 1-19.

Lister, D. B., R. J. Beniston, R. Kellerhals, and M. Miles. 1995. Rock size affects juvenile salmonid use of stream bank riprap. Pages 621-632 in C. R. Thorne, S. R. Abt, F. J. B. Barends, S. T. Maynord and K. W. Pilarczyk, editors. River, coastal and shoreline protection: erosion control using riprap and armourstone. John Wiley and Sons, New York.

Micheli, E. R., J. W. Kirchner, and E. W. Larsen. 2004. Quantifying the effect of riparian forest versus agricultural vegetation on river meander rates, central Sacramento River, California, USA. River Research and Applications 20: 537–548.

Moyle, P. B., R. D. Baxter, T. Sommer, T. C. Foin, and S. A. Matern. 2004. Biology and population dynamics of Sacramento splittail (*Pogonichthys macrolepidotus*) in the San Francisco Estuary: a review. San Francisco Estuary and Watershed Science [online serial] 2: Article 3 [http://repositories/cdlib.org/jmie/sfews/vol2/iss2/art3].

NODOS TAG (North-of-the-Delta Offstream Storage Technical Advisory Group). 2004. Sacramento River flow regime: Summary report and evaluation. Prepared for North-of-the-Delta Offstream Storage Project Management Team. Administrative draft.

Peterson, D., G. H. Golet, R. Luster, M. Roberts, and D. Jukkola. 2003. Modeling plant community types as a function of physical site characteristics. Prepared by The Nature Conservancy, Chico, California for CALFED Bay-Delta Program, Sacramento, California.

Schumm, S. A. 1977. The fluvial system. Wiley-Interscience, New York.

Small, S. L., N. Nur, A. Black, G. R. Geupel, D. Humple, and G. Ballard. 2000. Riparian bird populations of the Sacramento River system: results from the 1993–1999 field seasons. Prepared by Point Reyes Bird Observatory, Stinson Beach, California for The Nature Conservancy and U.S. Fish and Wildlife Service.

Sommer TR, Nobriga ML, Harrell WC, Batham W, Kimmerer WJ. 2001. Floodplain rearing of juvenile chinook salmon: evidence of enhanced growth and survival. Can J Fish Aquat Sc 58(2):325–33.

SRCAF (Sacramento River Conservation Area Forum). 2003. Sacramento River Conservation Area Forum handbook. Prepared for The Resources Agency, State of California by the Sacramento River Advisory Council; revised and updated by the Sacramento River Conservation Area Forum, Red Bluff, California. http://www.sacramentoriver.ca.gov.

The Resources Agency. 1989. Upper Sacramento River fisheries and riparian habitat management plan. Prepared for The Resources Agency, State of California by Upper Sacramento River Fisheries and Riparian Habitat Advisory Council.

TNC (The Nature Conservancy). 2005. Restoring Biodiversity Health on the Sacramento River. Prepared by The Nature Conservancy, Chico, California for CALFED Bay-Delta Program, Sacramento, California.

Trush, W. J., S. M. McBain, and L. B. Leopold. 2000. Attributes of an alluvial river and their relation to water policy and management. Proceedings of the National Academy of Sciences 97: 11858–11863.

USACE (U.S. Army Corps of Engineers). 1999. Post-flood assessment for 1983, 1986, 1995, and 1997 in the Central Valley, California. USACE, Sacramento and San Joaquin River Basins Comprehensive Study, Sacramento District, Sacramento, California.

USACE (U.S. Army Corps of Engineers). 2004. Hamilton City flood damage reduction and ecosystem restoration, California. Final feasibility report and Environmental Impact Statement/Environmental Impact Report USACE, Sacramento, California.

USDA Forest Service. 2005. Upper Trinity River watershed analysis. USDA Forest Service, Shasta-Trinity National Forest, Redding, California.

USFWS (US Fish and Wildlife Service). 2000. Impacts of riprapping to ecosystem functioning, lower Sacramento River, California. Prepared for Sacramento District, US Army Corps of Engineers, for the USFS Wildlife Coordination Act Report and Biological Opinion for proposed bank protection work.

Upper Sacramento River Salmon and Steelhead Advisory Committee (USRSSAC). 1983. Report No. 1: Red Bluff Diversion Dam and Tehama-Colusa Fish Facility July 1983.

Vogel, D. A., K. R. Marine, and J. G. Smith. 1988. Fish passage action program for Red Bluff Diversion Dam. Final report on fishery investigations. FR1/FAO-88-19. U.S. Fish and Wildlife Service, Fisheries Assistance Office, Red Bluff, California.

Figures

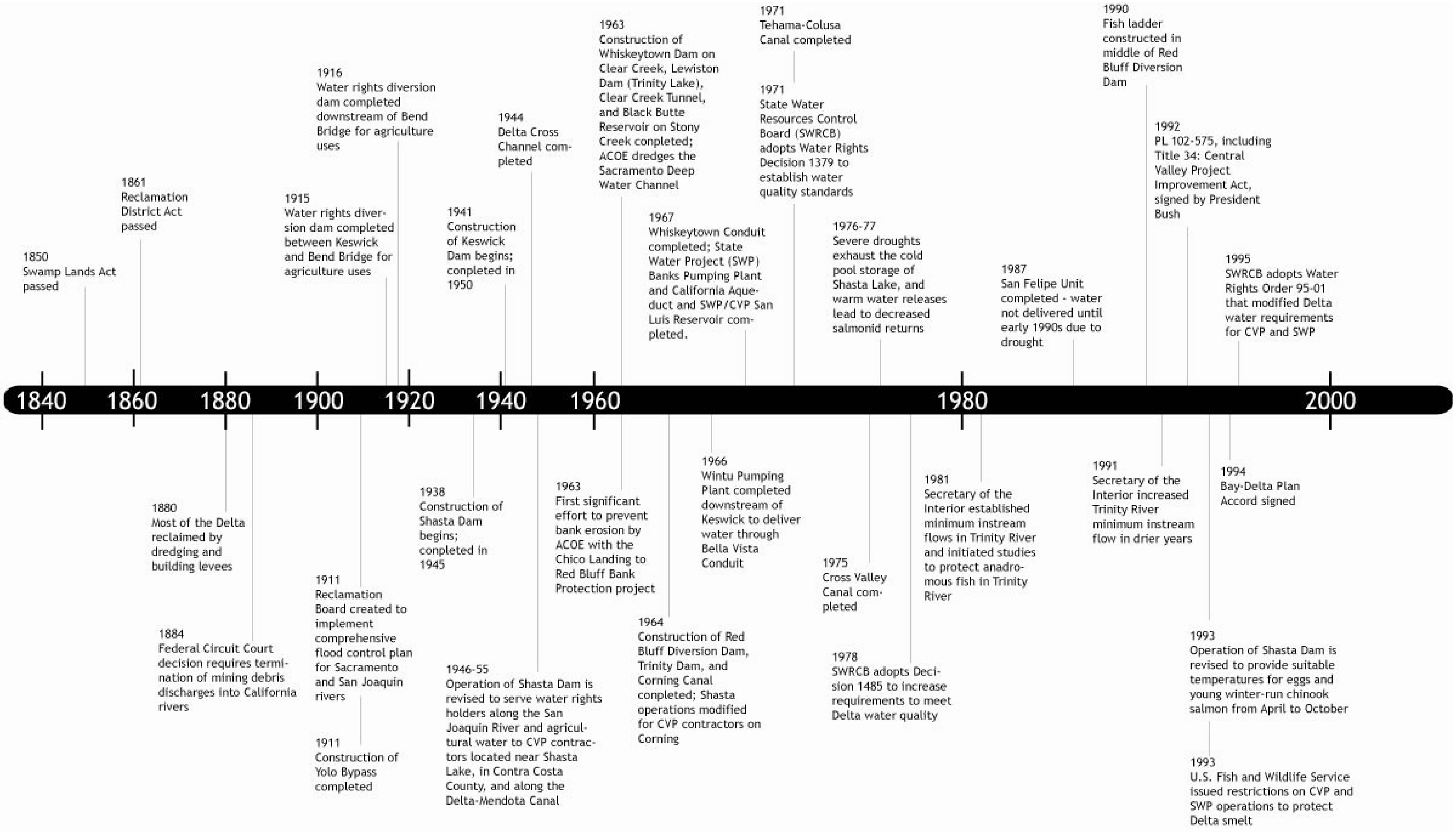


Figure 2-1. Timeline of human activities that led to major changes in ecological processes and watershed inputs of the Sacramento River system.

Figure 2-2. Lasting effects of aggregate extraction. Kutras Park (center) was one of the primary sources of aggregate for construction of Shasta and Keswick dams. Overall, more than 7 million cubic yards of sediment has been mined from the upper Sacramento River basin. Much of the mining has been done on the mainstem channel and floodplain. Remnant mining pits can trap sediment and disrupt the continuity of bedload transport. (Source: CDWR, 1999. Sacramento River Aerial Atlas.)

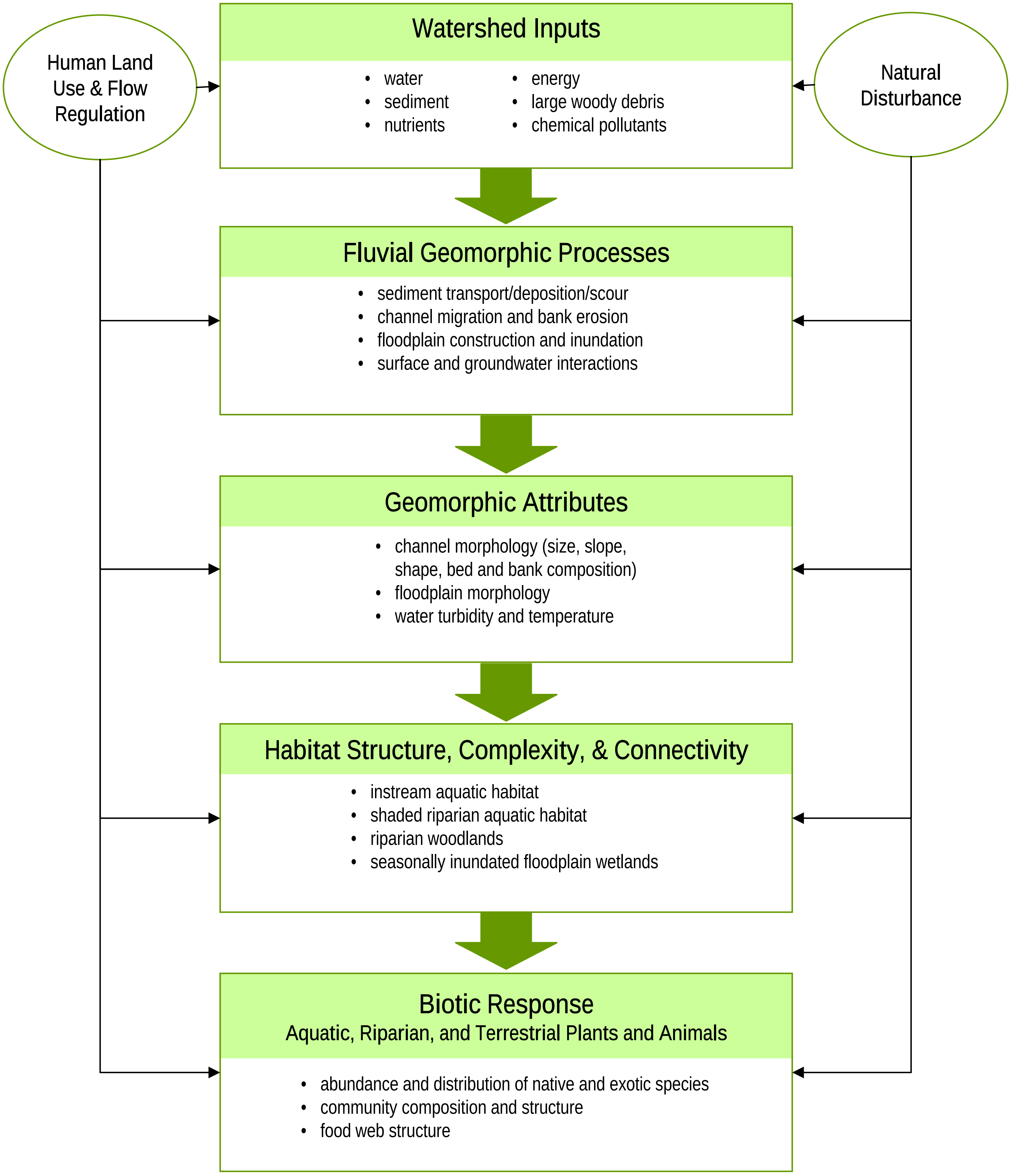


Figure 2-3. A simplified conceptual model showing the linkages between watershed inputs, fluvial geomorphic processes and attributes, habitat conditions, and the biota of river-floodplain-riparian ecosystems.

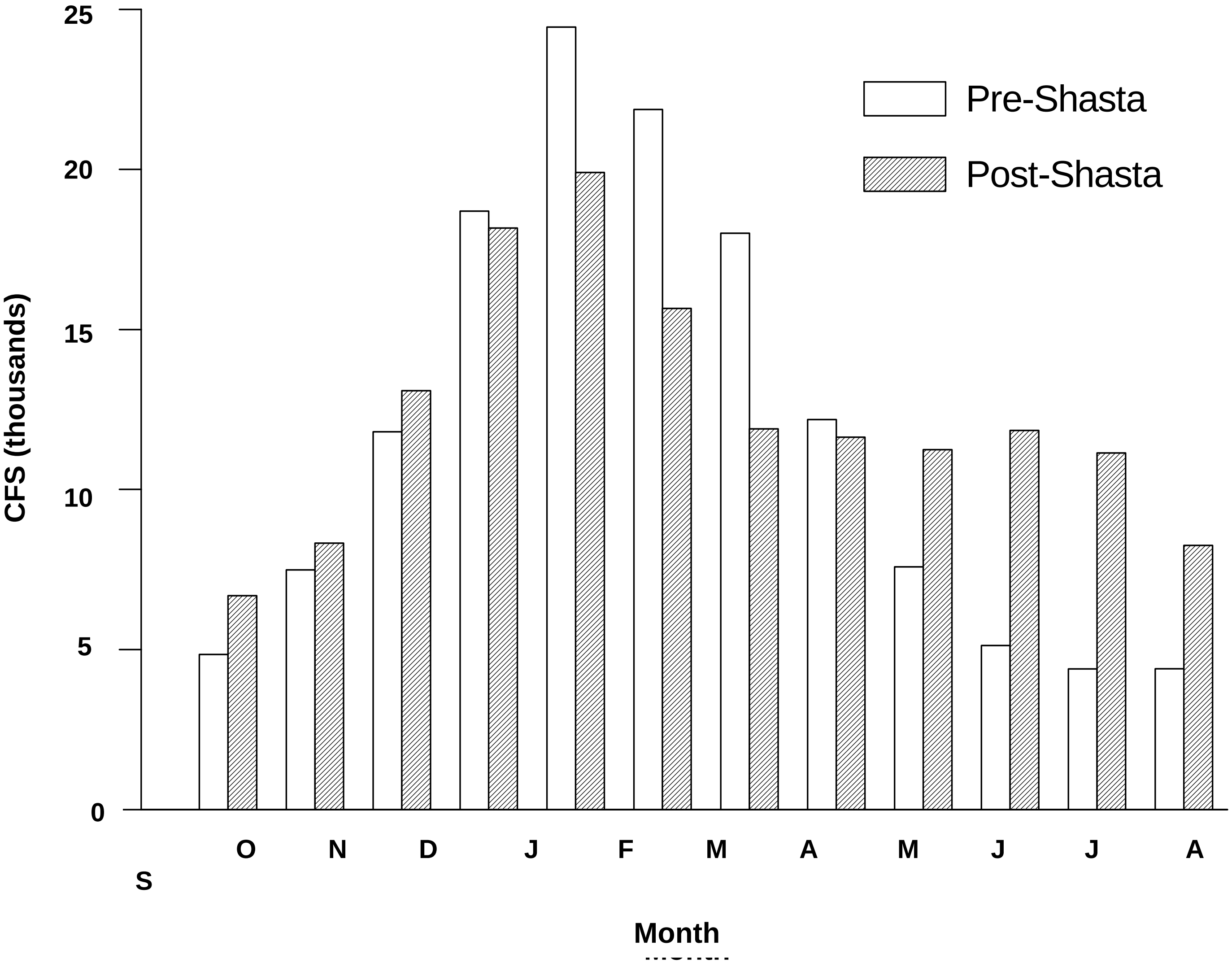


Figure 2-4. Mean monthly discharge at the Bend Bridge gauge (USGS # 11377100) before and after the construction of Shasta Dam. In general, Shasta Dam operations have reduced winter flows and increased summer and fall baseflows as a function of storing winter rains for delivery as irrigation supply during the growing season. Source: Kondolf et al. 2000.

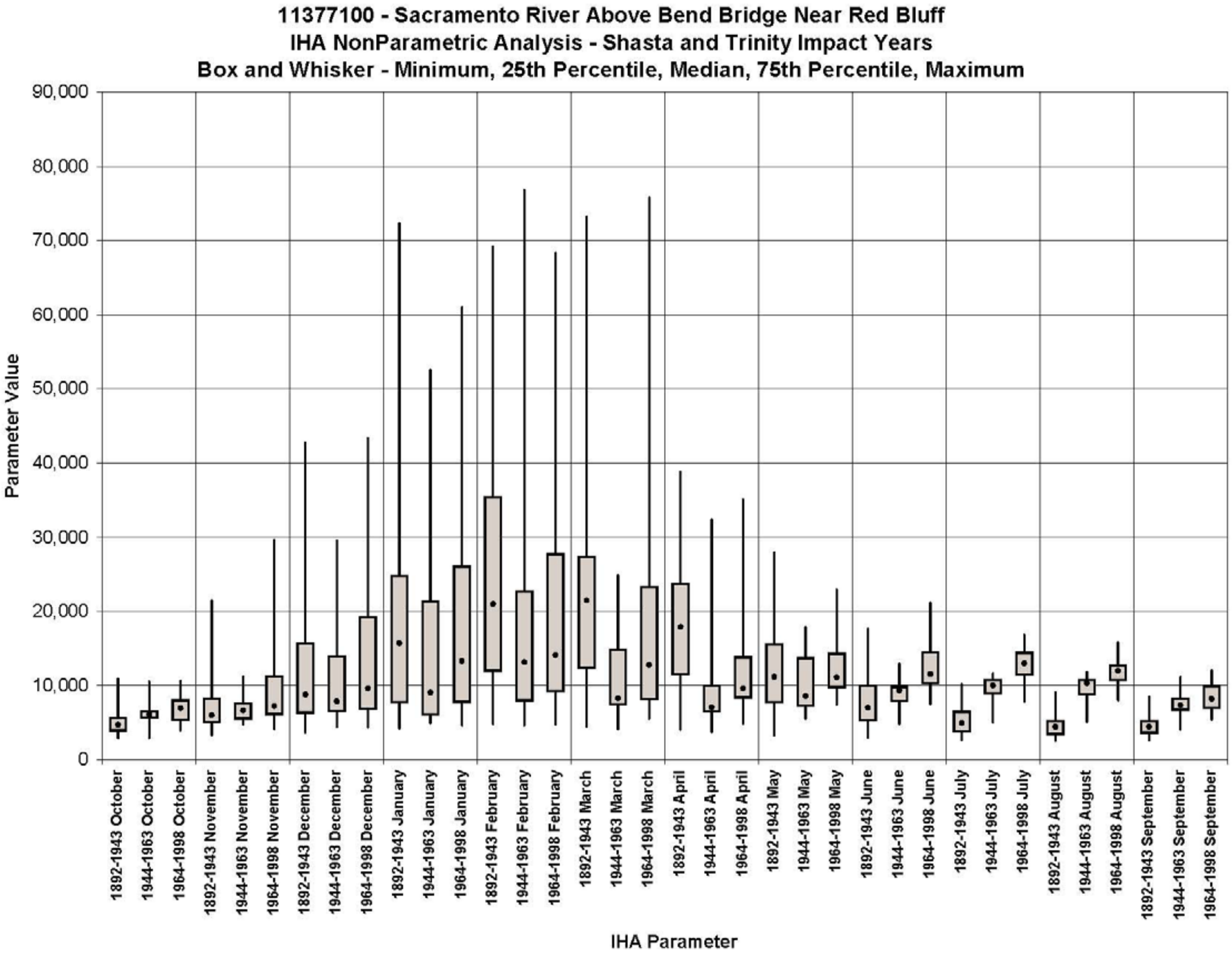


Figure 2-5. Mean monthly discharge at the Bend Bridge gage for three time periods: pre-Shasta (1892−1943), post-Shasta/pre-Trinity diversion (1944−1963), and post-Shasta/post-Trinity (1964−1998). The solid dot represents the median value, the shaded box delineates the 25th and 75th percentiles, and the thin lines (whiskers) delineate the minimum and maximum values. Source: CDWR 2001, as reported in Roberts et al 2002.

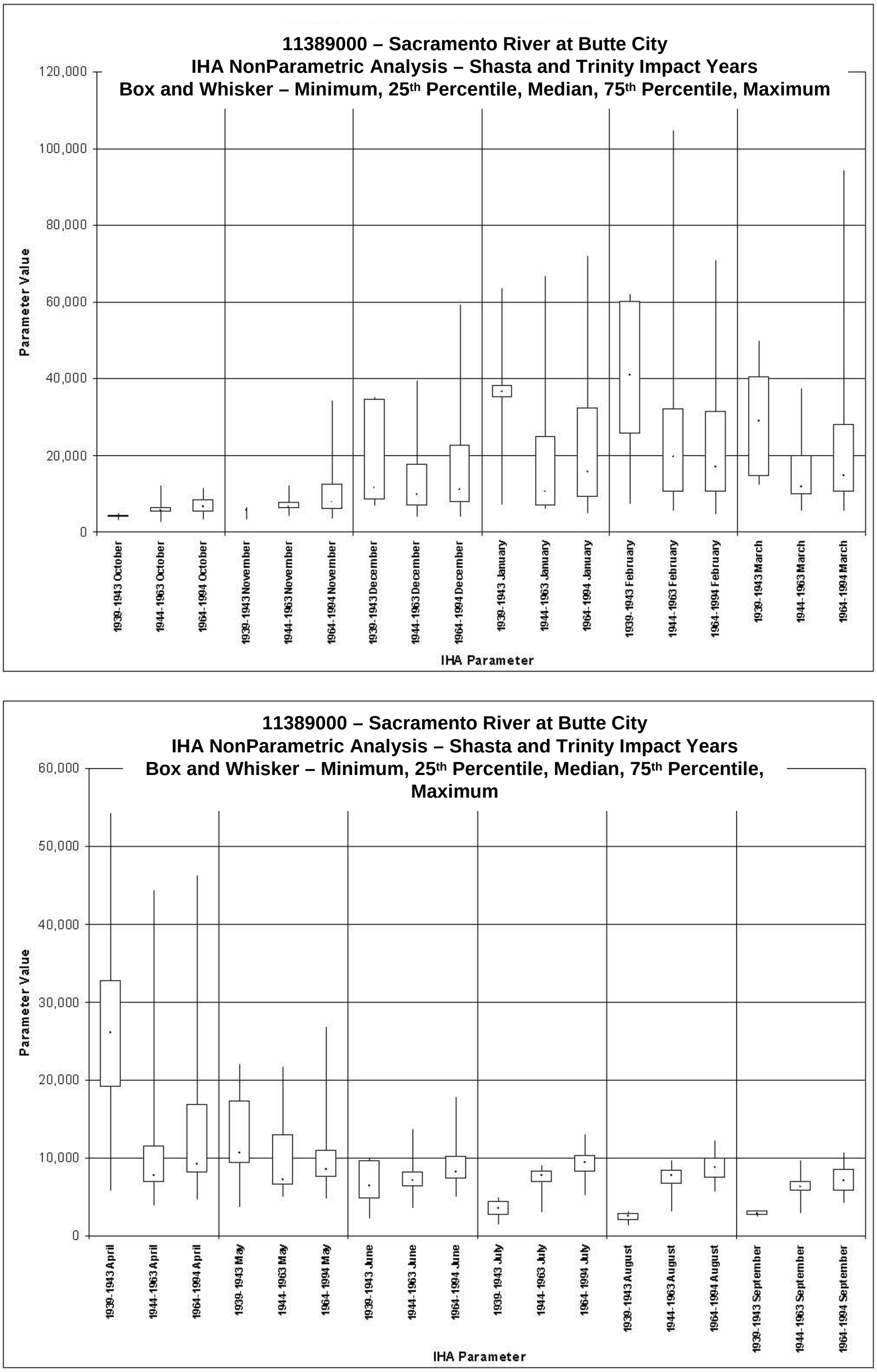


Figure 2-6. Comparison of mean monthly discharge at the Butte City gage for three time periods (see Figure 2-5 for explanation of time periods and box and whisker plots). Note the "flow reversal" pattern with increased average monthly flows under regulated conditions in summer and reduced flows in winter months. Source: CDWR 2001, as reported in TNC 2003.

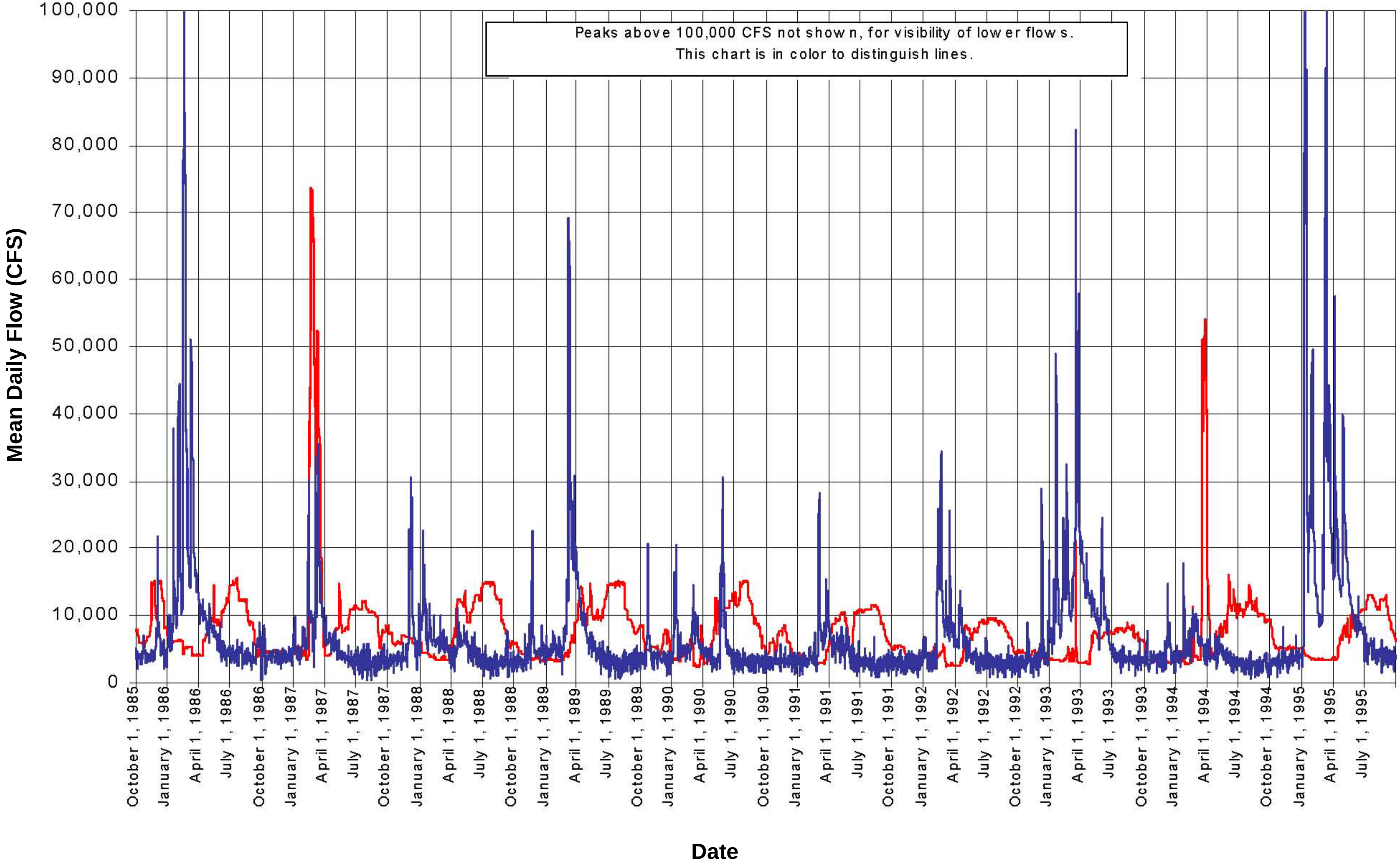


Figure 2-7. Comparison of Sacramento River inflow to Shasta Reservoir (blue) with outflow below Keswick Dam (red), based on mean daily flow for Water Years 1985–1994. Source: Roberts et al. 2002.

3 GEOMORPHIC CONTROLS ON HABITATS OF SACRAMENTO RIVER FOCAL SPECIES

Riverine and floodplain habitats are created, maintained, and destroyed by geomorphic processes whose rates and patterns are regulated through complex interactions of flow, sediment transport, and the properties of the channel and floodplain (including slope, erodibility, and morphology). Because large systems such as the Sacramento River and its floodplain generally support a diverse array of habitats and species and are affected by the interaction of a wide variety of geomorphic processes, quantifying and understanding how they evolve can be problematic. The legacy of land and water use in a region add to the complexity, modulating factors such as flow, sediment supply, and floodplain erodibility, and thus affecting the dynamics of riverine and floodplain habitats.

The effects of management decisions on physical parameters (such as the magnitude and frequency of peak flow, for example) can often be quantified more or less straightforwardly (see examples in Chapter 2). The implications for geomorphic processes and habitat dynamics are conversely much more difficult to determine, because relationships between process and form for channels and floodplains are typically complex and therefore not always easy to understand. The complexity of the Sacramento River in particular was highlighted in a recent analysis of flow and ecological processes (Kondolf et al. 2000). The analysis identified limitations and gaps in existing data and models. Of particular concern, according to the analysis, are uncertainties in estimates of sediment supply and the magnitude, timing, and duration of peak flow (Kondolf et al. 2000), which together are the fundamental regulators of sediment mobilization, bed scour, riparian recruitment, and bank erosion.

To the extent permitted by available data and analyses, this chapter describes relationships between riverine process and habitats of the Sacramento River focal species. Discussion of how rates of sediment supply and transport affect grain-size distributions and topographic characteristics of riffles (presented in Section 3.1), for example, is relevant to understanding the distribution, quality, and dynamics of spawning habitat for Chinook salmon (Chapter 4), steelhead (Chapter 5), and green sturgeon (Chapter 6). Discussions of meander migration processes and point bar formation (Section 3.2) are relevant to bank swallows (Chapter 7), western pond turtles (Chapter 8), and riparian succession (Chapter 9), which variously rely on main-channel, streamside, and off-channel aquatic habitats that are created and maintained as a function of the local rate, pattern, and style of lateral channel migration. We conclude each section (i.e., 3.1 and 3.2) by proposing several geomorphic metrics of ecosystem health and discussing how they may have changed over time in response to variations in flow and sediment transport.

An exhaustive annotated review of geomorphic studies of the Sacramento River is beyond the scope of this SOS Report. In keeping with the overall approach, this chapter focuses on how the formation, maintenance, and destruction of focal species habitats are affected by geomorphic processes. In this way, a foundation is laid for Chapters 4 and 9, where habitat requirements are presented in detail and where conceptual models of habitat dynamics provide context for understanding how human activities have affected geomorphic processes and thus altered habitats on the Sacramento River.

3.1 Key Habitat Types

One goal of the focal species selection process was to define a suite of species that covered a broad range of habitats that occur in the Sacramento River corridor, which expands the relevance of the discussion beyond the focal species to cover other species that use the same habitat types. This section provides a brief description of some of the key habitat types in the Sacramento River, supplemented by the identification of which focal species use each habitat type. By briefly describing and illustrating key habitat types, this section establishes a glossary of habitat units for the remainder of the report. The remainder of Chapter 3 discusses the ecological processes that help to form and maintain these habitat types.

3.1.1 Gravel-bedded riffles

Salmonids generally spawn in areas where the bed is composed of gravel and hydraulic conditions provide appropriate water depths and flow velocities. High flow events can mobilize and scour gravel stored in the channel bed, routing the sediment downstream. In the alluvial reaches of unregulated rivers, the sediment that is scoured from a local reach is generally replaced by sediment that is transported from upstream, supplied from tributaries, or recruited from storage in river banks. There may be short-term or local changes in the amount of gravel stored in a channel bed because of episodic sediment delivery (e.g., mass wasting events in the watershed) or extreme flow events, but over a broader time span, unregulated rivers generally achieve a balance between sediment supply and routing so that in-channel sediment storage is maintained.

3.1.2 Point bars

Point bars generally form on the inside of meander bends, where hydraulics cause velocities to be slower as compared to the outside of meander bends. As a result, gravel deposits on the inside of a bend, usually in arcuate forms. Point bars often provide geomorphic surfaces for riparian vegetation (e.g., cottonwood) colonization. Aquatic zones around point bars can also provide mainstem rearing habitat for salmonid fry.

3.1.3 Cutbanks

High flows can erode banks, especially on the outside of meander bends where channel hydraulics cause flow velocities to be higher, as compared to the inside of meander bends. Bank erosion, in general, can recruit gravel stored in floodplains, which can be an important source in regulated systems where coarse sediment has been reduced by dams. In some locations, bank erosion can create vertical cutbanks that are several meters in height. Cutbanks are especially important for bank swallows, which nest in cutbanks located within zones of appropriate soils. Over time, cutbanks can slump and become less vertical as toes are eroded. The cutbank erosion process also serves as a mechanism for recruiting large woody debris into the river channel, which can provide important aquatic cover for salmonids and other fishes.

3.1.4 Pools

Pools are areas where, if discharge were reduced, there would be areas in the bankfull channel where water would pond, though the areas would be disconnected from mainstem flow because of a hydraulic control point. Pools often form on the outside of bends, where comparatively higher flow velocities can induce deeper scour of the bed surface. Pools can provide important holding habitat for adult winter-run and spring-run Chinook salmon, and for green sturgeon adults.

3.1.5 Eddy-point bar complexes

Eddy-point bar complexes are small channels that are scoured on the downstream end of point bars. They provide seasonally inundated aquatic habitat within the bankfull channel, which can be important rearing areas for juvenile salmon; however, they may also pose a stranding risk to juvenile salmon if reductions in discharge quickly disconnect the eddy-point bar complexes from the mainstem channel.

3.1.6 Side channels

Side channels are often former channel alignments that once conveyed at least a portion of flow. Side channels are generally abandoned by channel avulsion or by the transformation of an anabranching river to a single-thread meandering channel as a result of changes in flow and sediment supply. Side channels can often be distinguished from oxbow lakes by of the angle of approach of the mainstem channel relative to the former channel, with oxbow lakes having higher angles and side channels having lower angles. Side channels can provide rearing habitat for juvenile salmonids and ephemeral aquatic habitat for western pond turtle. Side channels are important recruitment zones for riparian vegetation including cottonwoods, which provide shaded riverine aquatic (SRA) habitat and vegetation complexity.

3.1.7 Oxbow lakes

Oxbow lakes form when meander loops are cutoff from the main channel, either as neck or chute cutoffs. Neck cutoffs occur when a meander bend has become so narrow that only a thin neck of land separates the upstream and downstream end of a meander loop, such that bank erosion eventually erodes the neck of land to carve a new mainstem channel that is shorter. Chute cutoffs occur when overbank flows carve a new channel through the floodplain that is enveloped on the inside of a meander loop. Eventually, the new channel carved on the floodplain captures the flow and becomes the mainstem channel, and over time, the upstream and downstream ends of the former meander bend are plugged with sediment, thereby isolating the old river channel from the mainstem channel. Oxbow lakes may provide the primary aquatic habitat for western pond turtle. They can also be havens for non-native bass species such as largemouth bass, an important salmonid predator. Generally, they serve as juvenile salmonid population sinks, because hydraulic connectivity with the mainstem channel generally doesn't occur with sufficient duration to facilitate egress (M. Limm, personal communication, May 1, 2005).

3.1.8 Inundated floodplains

Periodic high flow events spill from the bankfull channel to inundate the floodplains that border the Sacramento River. By connecting the floodplain with the channel, high flow events can stimulate nutrient and sediment exchange between the two zones. Inundated floodplains can expand rearing habitat for juvenile salmonids, and may offer better rearing conditions than are found in the mainstem channel because greater food availability and reduced predation pressure (Sommer et al. 2001a, 2001b). Greater food availability results from high levels of primary productivity that is found in these areas of shallow water and higher residence time. Floodplain inundation can also promote riparian vegetation colonization by creating moist, bare mineral soils that result from the deposition of fine sediment load as water velocities slow and promote seed establishment.

3.1.9 Summary of key habitats used by each focal species

Each of the six focal species uses one or more of the key habitats described above (Table 3-1). An overview of anthropogenic alterations affecting these habitats is provided in Chapter 2, while Chapter 3 describes the physical processes that create and maintain these key habitats. More details on how each focal species relies on these key habitats are provided in Chapters 4–9**.**

Table 3-1. Matrix indicating key habitats used by each focal species.

Focal species	Habitat type							
	Gravel-bedded riffles	Point bars	Cutbanks	Pools	Eddy-point bar complexes	Side channels	Oxbow lakes	Inundated floodplains
Chinook salmon	X	X		X	X	X		X
Steelhead	X	X		X	X	X		X
Green sturgeon	X			X				
Bank swallow			X					
Western pond turtle				X		X	X	X
Fremont cottonwood		X				X	X	X

3.2 Transport and Dynamics of Channel Bed Materials

The supply, transport, and size of sediment in rivers are the key regulators of the spatial distribution, grain size, and dynamics of riverine gravel deposits—which provide key habitat for many aquatic species. The linkages among physical processes, habitat, and biota in gravel deposits are therefore important to document and understand. The linkages are relatively well understood for anadromous salmonids, making them an ideal focal point for the following discussion about gravel dynamics.

3.2.1 Gravel and anadromous salmonids

The mainstem Sacramento River currently supports spawning of fall-run, late-fall-run, and the federally endangered winter-run Chinook salmon (*Oncorhynchus tshawytscha*), in addition to steelhead (*Oncorhynchus mykiss*). Self-sustaining populations of spring-run Chinook salmon once spawned throughout the Sacramento River system, but their continued existence on the mainstem has not been confirmed in recent studies.

All of the above species and races are considered in the focal species discussions in this report (see Chapters 4–5). The continued existence of anadromous salmonids in the Sacramento River system depends in part on the long-term fate of suitable spawning gravels. The extent, distribution, and quality of salmonid spawning habitat in rivers are determined by several factors, including the quantity and grain-size distributions of gravel in riffles.

Besides affecting the earliest life-stages of salmonids, the dynamics of gravel deposits also affect juvenile salmonids by affecting point bars and associated eddies, which provide rearing habitat.

Juvenile salmonids are also affected by the frequency of bed mobilization, which influences the types and abundance of aquatic macroinvertebrates, an important food source. These linkages are discussed in greater detail in Chapters 4 and 5. In the text that follows here in Chapter 3, we focus primarily on dynamics of gravel as they relate to salmonid spawning.

There are several suitability criteria for grain-size distributions of spawning gravel. First is an upper limit on particle size. Just one particle can render a deposit unspawnable, if it is so big that spawning fish cannot build a redd around it. Second is an upper limit on the overall percentage of excessively coarse material. If material that fish can not move is abundant it may be impossible to build a redd within it. The upper limit on particle size and percentage of coarse material are dictated by (1) the size of the spawning fish and (2) local hydraulic conditions including velocity and slope. Because local hydraulic conditions can vary substantially from point to point, it is difficult to quantify an upper limit to the size of spawning gravel in the Sacramento River.

The suitability of gravel for spawning is also governed by the percentage of excessively fine (< 0.08 in [< 2 mm] diameter) material (McCuddin 1977, Reiser and White 1988). As the concentration of fine sediment in the subsurface increases, there are important implications for the survival of salmonid eggs and alevin in spawning redds. Survival from egg incubation through fry emergence for salmonid fish depends on the presence of cool, clean intragravel flow, in quantities sufficient to ensure adequate delivery of dissolved oxygen and removal of metabolic wastes. When fine sediment becomes heavily concentrated in (or on) a streambed, the rate of intragravel flow in the substrate can be substantially reduced, due to a reductions in gravel permeability. The presence of abundant fine material can thus result in increasingly depleted dissolved oxygen concentrations (and increasingly elevated metabolic waste levels) around incubating eggs, larvae, and sac-fry as they develop within egg pockets (Kondolf 2000). This can lead to high mortality. Abundant fine sediment around egg pockets can further increase mortality of salmonid fry via entombment (i.e., when fine sediment plugs interstices such that fry cannot emerge). As a general rule, spawning salmonids require gravel that is "clean" (i.e., free of abundant fine material) (Kondolf 2000).

Spawning-sized gravel is typically mobilized in the spawning reaches of the Sacramento River by floods that have low-frequency and high-magnitude. Sand and fine sediment can be mobilized by much more frequent, lower magnitude flows (e.g., Knighton 1984). For the upper Sacramento River near Redding, it is probably safe to assume that sand is mobilized during all but the lowest flows. When mobilized, grains of sand tend to saltate (i.e., hop) along the bed and can eventually infiltrate into the interstices between coarse particles, which form the framework of the channel bed.

The concentration of fine sediment in the channel bed can be reduced periodically by flow events that are big enough to scour the subsurface and thus expose its fine sediment to downstream transport. Fine sediment can also be cleaned from the subsurface by adult salmon during redd construction, when they kick fine material into the water column with their tails and thus entrain it in downstream flows (Kondolf 2000). If flow is locally slow, such that sand doesn't travel very far downstream during redd construction, female salmon may inadvertently contaminate their redds with fines as they cover their eggs with sediment from upstream of the egg pocket. Moreover, if the concentration of fine sediment in the bed is too high, it can render the effects of gravel cleaning by salmon insufficient for survival of buried eggs.

3.2.2 Factors affecting the evolution of spawning gravel on the Sacramento River

Below we summarize relevant available data and analysis on the evolution of spawning gravel in the mainstem Sacramento River bed. The review is limited by the fact that historical grain size data date back to only 1980 and are only available for the mainstem from a few sampling efforts of sometimes limited spatial extent (CDWR 1980, 2002; Buer 1984, 1995; USACE 1981; RCE, 1992; WET 1988). Additional grain size data are available for banks (WET 1988; Buer 1994a, 1995; Klinesteker 1998) but are not reviewed here because they are not directly relevant to the evolution of spawning gravel in the mainstem channel bed.

3.2.2.1 Supply and transport of spawning gravel

Suitably sized spawning gravel deposits on the mainstem extend, in a disconnected patchwork, from Keswick Dam, which prohibits upstream fish passage at RM 302, to Colusa at RM 143, in the gravel-sand transition zone, where bed material grades from gravel to sand. However, the lowermost fall run spawning occurs near Princeton (RM 164) due to limitations imposed by water temperature. Additional discussions of suitability criteria for Chinook and steelhead spawning gravel are provided in Chapters 4 and 5.

High flow events can mobilize a channel bed, scouring gravel and transporting it downstream. In alluvial reaches of unregulated rivers, local scour is generally offset over the long term by deposition of sediment from tributaries and mainstem bank erosion. In-channel sediment storage generally remains constant over the long-term, although episodic sediment delivery (e.g., from landslides upstream) and extreme flow events can increase or deplete sediment storage in channels over the short term.

On the Sacramento River, the construction and operation of Shasta (RM 312) and Keswick (RM 302) dams have altered mainstem flow and sediment supply. This has affected the quantity and grain-size distributions of gravel stored in the downstream channel. This in turn has altered the extent and quality of salmonid spawning habitat.

Prior to the construction of the dams in the 1940s, the Sacramento River headwaters above RM 312 yielded an estimated average coarse sediment load of approximately 50,000 yd^3 (38,000 m^3) per year (K. Buer, pers. comm., 2005), implying a cumulative deficit since dam construction (i.e., over the past roughly 60 years) has been approximately 3 million yd^3 (2.3 million m^3).

The flow threshold for spawning gravel mobilization in the channel immediately below Keswick Dam has been estimated to be 50,000 cfs (CDWR 1981). This is considered to be a minimum estimate because it was based on observations of injected gravel that may have suffered preferential scour due to (1) its position relative to the high-velocity core of the flow and (2) the fact that it was not integrated into the framework of the bed. The 50,000 cfs mobilization threshold is only considered applicable to the reach (RM 302 to roughly RM 298) immediately below the dam. There have been several flow events with magnitudes greater than 50,000 cfs since the completion of Shasta Dam in 1945 (Figure 3-1), but dam operations have clearly reduced the frequency of high, bed-scouring flows (see Chapter 2). For high releases that do occur, the lack of sediment supply from the upper watershed has presumably increased entrainment of coarse sediment from the channel bed below the dams (e.g., Ligon et al. 1995).

Coarse sediment scour from the bed of the upper Sacramento River may have started even before Shasta Dam was completed in 1945. Construction-related coffer dams probably trapped sediment from the upper watershed, such that high flow events of 1940 (186,000 cfs), 1941 (82,300 cfs), and 1942 (85,000 cfs) (measured at the historical Kennet gauging site) were sediment-starved and may have scoured gravel stored in the channel bed below the dam site (K. Buer, pers. comm., 2005).

Dam-related reductions in sediment supply have been exacerbated by aggregate mining. An estimated 7.1 million yd^3 (5.4 million m^3) of sediment was removed from the upper Sacramento River (in the immediate vicinity of Redding at RM 298) for construction of the dams and related infrastructure (CDWR 1980). Remnant mining pits continue to affect the system by disrupting the continuity of sediment transport, trapping bedload as it is delivered by flow from upstream (Figure 2-2). Ongoing in-stream mining in tributaries (e.g., Clear Creek, Cottonwood Creek, Reeds Creek, Red Bank Creek, Stony Creek, and Thomes Creek) also affects coarse sediment supply to the mainstem (Buer 1994b).

The first significant natural source of sediment to the Sacramento River is nearly 30 mi (48 km) downstream of Keswick Dam at Cottonwood Creek (RM 273.5). Tributaries between Keswick Dam and Cottonwood Creek contribute little sediment to the mainstem, because they drain small basins of erosion-resistant material or, as is the case in particular for Clear Creek, are themselves regulated by dams and are affected by aggregate mining (CDWR 1980). Much of the upper Sacramento River (i.e., from RM 302 to approximately RM 273.5) is bounded by erosion-resistant bedrock and terrace deposits (CDWR 1980), such that bank erosion is not fast enough, relative to in-channel transport, to provide a significant source of coarse sediment. In other words, the rate of supply from erosion of banks due to meander migration in the upper river is minimal.

Without a supply of spawning-sized gravels to replenish material scoured and routed downstream by post-dam flow releases, the channel bed surface between Keswick Dam (RM 302) and Cottonwood Creek (RM 273.5) has presumably become progressively coarser, as large particles, which cannot be mobilized by the flow releases, have been left behind in armor-like lag deposits on the bed surface. In many locations, spawning-sized gravels in the subsurface (CDWR 1980) may not be available to salmon because they cannot mobilize the surface armor.

The hypothesis that dam- and mine-related surface coarsening has affected spawning habitat is supported by surveys of spawning habitat conducted by the California Department of fish and game (CDFG) and CDWR in 1964 and 1980. In particular, CDWR (1980) documented a loss of more than 50% of spawning habitat in the key spawning reach between ACID Dam (RM 298.4) and the City of Anderson (RM 283) (Figure 3-2). It is difficult to attribute this reduction to any one cause. For example, in the interval between the surveys, blockage caused by Red Bluff Diversion Dam at RM 243 (beginning in 1967) and changes in escapement may have contributed to changes in the distribution of spawning habitat mapped in the surveys. A difference in the resolution of the two habitat surveys is another potentially confounding factor (CDWR 1980). Despite the complications and confounding factors, CDWR (1980) concluded that the changes in spawning habitat between the two surveys indicated a loss of habitat that could be attributed at least in part to the effects of bed coarsening in the reach (RM 302–273.5).

Because there is little sediment input from tributaries on the upper Sacramento River, bed coarsening is expected to propagate downstream with successive high flow events. Sediment scoured from upstream reaches should initially provide supply for downstream reaches. Over time, however, as in-channel storage is depleted and gravels are trapped beneath an armor layer in upstream reaches, downstream reaches will lose their sediment supply and eventually become armored as well.

The confluence with Cottonwood Creek defines the downstream limit of plausible bed coarsening in the Sacramento River due to the locally high sediment supply which should induce fine sediment deposition (CDWR 1980). The time series of mapped spawning habitat area (Figure 3-2) corroborates this expectation. In the reach immediately above Cottonwood Creek (RM 273.5–276), there was a significant (nearly 50%) reduction in mapped spawning habitat between 1964 and 1980. Conversely, the reach below Cottonwood Creek (RM 269–273.5) showed comparatively little change in spawning habitat between the two surveys. Taken together the available evidence suggests that sediment supply from Cottonwood Creek helps maintain spawning habitat in the immediate vicinity of the confluence, despite reduced in-channel supply related to blockage of sediment from the upper watershed.

Bed coarsening has presumably been mitigated by the infusion of approximately 242,000 yd^3 (185,000 m^3) of spawning-sized gravel between 1978 and 2000 (Table 3-2). However, the total volume of added gravel has been small relative to the cumulative deficit of sediment since the construction of Shasta Dam. Even so, the added gravel has probably enhanced existing spawning habitat in the key winter-run spawning reach (RM 289.2–302), at least compared to what it would have been if no gravel had been added. Augmentation-related changes in spawning habitat are difficult to assess, because the vast majority of added gravel (i.e., nearly 95%) was injected after 1980, when the last spawning survey was conducted. Pre-1980 additions were small (just 13,000 yd^3 [10,000 m^3]) and confined to a short stretch of river near the Redding Riffle (RM 298). Hence, effects of the earlier augmentation projects on spawning habitat area were probably too small to be detectable in a comparison of data from the 1964 and 1980 habitat surveys.

Table 3-2. Timing, location, and quantity of injected spawning gravel.

Time frame	Location (RM)		Number of sites	Volume added	
	Upstream	Downstream		yd^3	m^3
1978–1980	298.3	297.7	3	13,300	10,169
1980–1995	302.0	290.0	9	123,910	94,736
1995–2000	302.0	291.6	3	105,366	80,558
Grand total	*302.0*	*290.0*	*15*	*242,576*	*185,462*

Source: CDWR (2002)

In general, it is difficult to know if the scale of recent gravel augmentation has had an appreciable effect on in-channel gravel storage or the extent of spawning habitat below Keswick Dam. The fact that > 10 million yd^3 (7.6 million m^3) of sediment have been mined from the channel and floodplain or trapped by dams implies that the effects of the injected gravel may not have been significant. The legacy of mining pits, which can act as sediment traps for the infusion projects, adds further uncertainty to assessment of the benefits of the injection projects (Buer 1995). A case in point is the abandoned pit at Kutras Park (Figure 2-2), which, at RM 296, breaks the continuity of downstream sediment transport from many of the injection sites. An additional pit at Shea Levee (RM 290) has minimal effects on sediment transport continuity, because the mainstem flow is directed around the pit by the levee (Buer 1994b).

3.2.2.2 Spawning gravel quality

In a 1995 gravel study of the upper Sacramento River between Keswick Dam (RM 302) and the confluence with Cottonwood Creek (RM 273.5), bulk samples were collected to characterize spawning gravel quality (Buer 1995). Results indicated that intra-gravel permeability was moderate to high in the mainstem channel bed above Cottonwood Creek (Buer 1995), implying that fine sediment concentrations were probably not a limiting factor for spawning in the reach. Several factors probably contribute to a relatively low concentration of fine sediment in the channel bed between Keswick Dam and Cottonwood Creek. First is the long post-dam period of clear-water releases, which have presumably depleted fine sediment from the channel bed by entraining it into the flow without bringing new additions from upstream as flows recede. Second is the effect of Shasta and Keswick dams, which have reduced the overall supply of sediment from the upper watershed. Third is the relatively high erosion-resistance of material that bounds the mainstem channel over much of the upper reach. On the other hand, recently tilled dirt of adjacent agricultural lands may contribute fine sediment to the river. The extent to which this is the case for the Sacramento River is currently unknown.

Concentrations of fine sediment in the Sacramento River bed are probably much higher downstream of RM 273.5, where high sediment supply from Cottonwood Creek provides for fine sediment accumulation in the bed. Fine sediment concentrations may also be relatively high in the reach immediately upstream of RM 273.5, due to deposition caused by backwater effects when Cottonwood Creek flow is high. Additional bulk samples from bars and riffles above and below Cottonwood Creek would help verify whether this is the case.

3.2.3 Geomorphic metrics of changes in spawning gravel quality and area

After considering the above conditions and processes in the upper Sacramento River, we have developed three working hypotheses about the dynamics of salmonid spawning gravel in the Sacramento River. These hypotheses form the core of a gravel study which is currently underway as part of the Sacramento River Ecological Flow Study (Stillwater Sciences 2005). Each hypothesis is based on a metric of geomorphic change: (1) bed coarsening over time, (2) the downstream propagation of bed coarsening, and (3) changes in fine sediment concentration over time. As discussed below, it should be possible to assess changes in these metrics through analysis and modeling of existing and new data on spawning gravel quality and area.

3.2.3.1 Hypothesis 1: Bed coarsening over time

Hypothesis 1 is that progressive coarsening of the bed has continued to reduce the extent of salmonid spawning habitat between ACID Dam (RM 298.4) and Anderson Bridge (RM 283). The hypothesis that the bed of the upper Sacramento River coarsened in the post-Shasta era is not new, and has been supported by results from several field studies (CDWR 1980, Buer 1995). For example, as noted in Section 3.1.2, comparison of the 1964 and 1980 spawning habitat maps suggests a significant loss of spawning habitat between RM 298.4 and RM 283. The remaining spawning habitat appears to be confined to relict features (such as point bars) in zones where local hydraulics prevent high flows from eroding gravels (Buer 1995). This is presumably due to bed coarsening related to reductions in sediment supply and storage in the upper Sacramento River. Since 1980 and 1995, more of the bed surface has presumably become armored, while increasing amounts of fine sediment may have become trapped in the subsurface of spawning areas.

The expectation that the surface has continued to coarsen while the subsurface has been contaminated by fine sediment is based on conventional understanding of sediment transport in reaches downstream of dams (Ligon et al. 1995). Whether this expectation is valid for the mainstem Sacramento River is unresolved. For example, a time series of Wolman pebble counts from bars and riffles in the upper Sacramento River shows that while surfaces at some sites became coarser, others became finer, or showed little change between sampling efforts in 1979–1980, 1995, and 2001 (Figure 3-3). Comparisons among bulk samples spanning the same interval (Buer 1995) provide an inconclusive test of coarsening, due to the small number of samples collected and inconsistencies in sampling methods for the two sampling efforts[2]. The ambiguity of the grain size results from 1980 and 1995 fails to support the coarsening hypothesis for post-1980 interval. The observed patterns might be explained by variability in grain-size distributions due to natural processes in the absence of coarsening over time. Moreover, temporal changes in grain-size distributions of point bars may not strongly reflect temporal variations in grain size of spawning patches. Hydraulics of point bars make them likely to receive much of the sediment that deposits in the receding stages of floods. As a result, they may be the last features to exhibit significant coarsening after a shutdown of sediment supply.

Natural processes contribute to spatial as well as temporal variability in grain-size distributions. Variability on a given point bar in the Sacramento River can be significant (CDWR 1980). Because successive sampling efforts were unable to collect grain-size data from exactly the same point, comparisons of grain-size data across the time series may reflect spatial variability more than temporal shifts, even if coarsening has been significant.

Alternatively variations in grain-size over time may reflect the effects of periodic gravel augmentation, which may have mitigated coarsening by supplementing in-channel sediment storage between RM 289.2 and RM 302. But the latter explanation seems unlikely for at least two reasons:

1. The total volume of augmented sediment in the 1980–1995 interval is 5 times lower than the supply would have been, based on the estimated pre-dam annual average coarse sediment load from the upper watershed. In general, the scale of gravel augmentation implemented to date (243,000 yd^3 [186,000 m^3]) is small compared to the estimated cumulative volume of coarse sediment retained by dams (> 3 million yd^3 [2.3 million m^3]), mined from the basin (> 7 million yd^3 [5.4 million m^3]), and scoured from the channel bed in the post-dam era. Recent gravel additions are probably too small to have appreciably reversed the cumulative loss of spawning habitat area in the upper Sacramento River.
2. Because manually added gravel is generally more susceptible to mobilization and transport than naturally deposited bed sediment (due to a more homogeneous grain-size distribution and lack of imbrication), the augmented gravel in all likelihood soon became widely dispersed downstream, such that it may currently have limited habitat value. This

---

[2] Bulk sampling methods in the 1980 and 1995 surveys were different: 12 in (30 cm) McNeil samplers were used in 1980, while 3 ft x 3 ft (0.9 x 0.9 m) plots were excavated by shovel in 1995 surveys. This produced a difference in the size (i.e., the overall mass) of the samples; in general, the 1995 samples were larger and thus statistically more robust as indicators of grain size. Moreover, the methods used in 1980 may have biased the sampling toward finer grain sizes, with the narrow gauge of the McNeil sampler preventing adequate sampling of coarse material. This bias would tend to produce the appearance of coarsening over time (Stillwater Sciences 2005). The coarsening reported by Buer (1995) in CDWR's comparison of bulk samples from 1980 and 1995 thus appears to be at least partly due to sampling biases.

is confirmed by observations of the 13,300 yd^3 (10,200 m^3) added to the upper Sacramento River in 1978 and 1979; an estimated 85% of it was eroded by high flows of 36,000 cfs and 50,000 cfs in the winter of 1980, while relatively little of the natural sediment at the same site was scoured by the same flows (CDWR 1980). Flows above the estimated 50,000 cfs spawning gravel mobilization threshold have been infrequent since augmentation began in 1978 (Figure 3-1), but have probably nevertheless been sufficient to mobilize much of the added gravel and transport it downstream.

Even if it could be shown that the coarse sediment augmentation program has mitigated the effects of coarsening over the recent past, it would be difficult to quantify the implications for spawning habitat, in the absence of a more recent spawning habitat survey (i.e., one that post-dates the 1980–2001 period of augmentation). Moreover, because the added gravel has been injected locally at a series of discreet sites separated from one another by large distances (in some cases many miles), any enhancements to spawning habitat have probably been localized to small areas downstream of the injection sites. This is supported by detailed inspection of the spawning habitat survey data from 1964 and 1980 (Figure 3-4). Spawning habitat near tributary confluences (e.g., Cow Creek and Stillwater Creek) remained stable or increased slightly between the surveys, highlighting the importance of small additions of gravel for local maintenance of spawning habitat. However, because the sediment loads supplied by tributaries between Keswick Dam and Cottonwood Creek were small and localized (Table 3-3), they appear to have had little effect on areas a few miles downstream, where spawning area was observed to decrease during the survey interval (Figure 3-4).

Table 3-3. Average annual sediment yields for the Sacramento River and major tributaries.

Location	RM	Drainage area		Bedload[1]		
		mi^2	km^2	Coarse gravel and coarser tons/year	Fine gravel and finer tons/year	Total bedload tons/year
Sacramento R., Keswick[2]	302.0	6,468	16,752	0[2]	0[2]	0[2]
Clear Creek	289.2	228	591	1,000	5,000	5,000
Churn Creek	284.6	12	31	1,000	3,000	4,000
Stillwater Creek	281.1	106	275	1,000	7,000	8,000
Cow Creek	280.1	684	1,772	2,000	17,700	19,700
Bear Creek	277.7	122	316	1,000	3,000	4,000
Battle Creek[3]	271.5	357	925	0[3]	0[3]	0[3]
Cottonwood Creek	273.5	927	2,401	3,000	17,000	20,000
Sacramento R., Bend Bridge	260.0	8,900	23,051	9,000	44,000	53,000
Reeds Creek	244.7	75	194	2,200	13,800	16,000
Red Bank Creek	243.3	94	243	2,700	16,300	19,000
Elder Creek	230.4	136	352	6,800	27,200	34,000
Thomes Creek	225.2	203	526	4,900	57,100	62,000
Mill Creek	230.0	208	539	1,900	500	2,400
Deer Creek	219.5	131	339	2,700	900	3,600
Sacramento R., Hamilton City	199.3	10,833	28,057	38,000	188,000	226,000

[1] In English tons/year. Note that the sum of inputs from above the mainstem locations (Bend Bridge and Hamilton City) do not add up to the estimated loads at the mainstem gauges. The data do not provide a balanced budget for coarse sediment. This is probably due at least in part to uncertainties and assumptions in the calculation methods. For

example the bedload from Thomes Creek was estimated to be 6% of the suspended sediment load at Paskenta. While such an approach may provide a reasonable first approximation, it cannot be expected to contribute to a balanced sediment budget for the mainstem.

2 Bedload is zero below Keswick Dam because it traps coarse sediment from the upper watershed.

3 Bedload from Battle Creek is assumed to be zero due to its low slope immediately upstream of its confluence with the Sacramento River; the mouth of the creek presumably acts as a coarse sediment trap that minimizes delivery to the mainstem over the short term (CDWR 1980). Over the long-term, the sediment load of Battle Creek is probably more substantial (as it must over time ultimately pass the sediment delivered to it from upstream sources).

Source: CDWR (1980) and Buer (1994a).

Recent gravel injections can be expected to have produced similar, mostly localized effects. Existing patches of spawning habitat near injection sites can presumably be maintained or supplemented when gravel is added. The effect is probably manifested in an increase in depth or area along habitat margins. However, as the added gravel is mobilized and transported farther downstream, it presumably becomes less useful, because it is more likely to end up widely dispersed in deep pools, thin lenses, and in areas that are not hydraulically suitable for spawning. Observations of broad areas of spawning-sized gravels in hydraulically unsuitable conditions (Buer 1995) lend support for this expectation. It is worth noting that some of the gravel added upstream of the remnant mining pits between RM 295 and RM 298 has probably become trapped in the pits. Nearly 25% of the gravel added below ACID Dam (RM 298.5) was placed upstream of pits at RM 295 (Buer 1995).

In summary, gravel augmentation and sediment supply from small tributaries appear to have had only local effects on spawning habitat. In-channel coarse sediment storage in reaches between the tributaries and injection sites has probably decreased since the 1980 spawning habitat survey due to bed-scouring flows.

3.2.3.2 Testing of hypothesis 1

Taken together, these considerations imply that the scale of gravel augmentation to date has probably provided short-term benefits for spawning habitat in the vicinity of injection sites, but have had little or no enhancement of spawning habitat at more distal sites downstream. This is an important part of hypothesis 1: that bed coarsening in the upper Sacramento River has continued and that spawning habitat has been further reduced in this reach, despite the effects of recent gravel augmentation.

A new spawning habitat survey, conducted as part of the ongoing Sacramento River Ecological Flows gravel study, will help test hypothesis 1. As part of the gravel study, the analyses of spawning maps from 1964 and 1980 are being revisited, with smaller spatial bins than the one-river-mile resolution shown in Figures 3-2 and 3-4, to better resolve the habitat losses in the intervening period. This should permit a more detailed assessment of factors influencing local changes in habitat area. For example, the effects of tributaries and gravel injection projects should be more clearly evident. In addition, the precise locations of remnant mining pits and deep pools will be assessed so that their sediment trapping effects can be better quantified and understood. The revised spatial analysis should thus help point to mechanisms underlying observed changes in spawning habitat and the grain-size distributions of channel bed materials.

The testing of hypothesis 1 will also be informed by analysis of existing and new grain size information. Previous analyses of existing grain-size data (CDWR 1980, 2002; Buer 1995) focused primarily on trends in indices of the distributions (e.g., D_g, D_{50} and D_{84}) rather than variations in their cumulative distribution functions. While indices such as D_{50} and D_{84} may sometimes be representative and instructive for analyses of changes in grain size from point to point and time to time, they may not always tell the whole story. For example, D_{50} can increase due to any one of many changes in the grain-size distribution. Scour of gravel (> 0.08 in [> 2 mm]) and scour of sand (< 0.08 in [< 2 mm]) could both reduce D_{50} while having nearly opposite effects on spawning gravel quality. Without more detailed information about individual grain-size distributions, it is difficult to determine the relative importance of the various mechanisms that could affect D_{50}. Analysis of the relative importance of each mechanism is important because it can provide a test of the bed coarsening hypothesis. If grain-size distributions grew coarser due to reductions in fine sediment, then the implications of increased D_{50} for spawning habitat might be minimal. If that is the case then it would further imply that differences in other factors (i.e., changes in upstream passage and differences in escapements, if significant) during the survey years may have been key regulators of the observed decrease in spawning habitat. Given the implications for increased intra-gravel flow, an increase in D_{50} due to decreases in fine sediment would actually be indicative of improved spawning conditions. Conversely, degradation in remaining spawning habitat would instead be implied if the increases in D_{50} primarily reflect the depletion of spawning-sized gravel from the channel bed.

It should be possible to further test the bed coarsening hypothesis through application of The Unified Gravel and Sand model (TUGS) to the upper Sacramento River. TUGS is a new sediment transport model that predicts changes in surface and subsurface grain-size distributions using data on flow, sediment supply, and the initial grain-size distributions of bed sediments. TUGS was developed to simulate the effects of different management actions (e.g., changes in the flow regime, gravel augmentation) on spawning habitat quality by predicting (1) the concentration of fine sediment in the channel bed and (2) reach-averaged values of grain size. The results of these simulations can be readily incorporated into the SacEFT of the Sacramento River Ecological Flows Study. As part of the gravel study currently being conducted by Stillwater Sciences, TUGS will also be used to help test the bed coarsening hypothesis, by simulating the evolution of the channel bed below Keswick Dam (i.e., between RM 302 roughly RM 243) following the construction of Shasta Dam. The simulation will be carried forward from 1945 (when Shasta Dam was completed), using existing slope and channel geometry data. The evolution of the grain-size distribution of the channel bed will be tracked as a function of the effects of historical flows (measured at USGS gauges), under the condition of zero sediment supply from the upper watershed. Because pre-Shasta Dam grain-size data for the channel bed are not available, a representative initial grain-size distribution will need to be assumed for the TUGS simulation. To help assess uncertainties introduced by this assumption, a statistically robust sensitivity analysis will need to be performed on the model results. The model results will be compared against measured grain-size distributions (from 1980, 1995, 2001, and the current gravel study) to see if there are any correlations with observed trends. Indications of increases in grain size over time will lend further support for the bed coarsening hypothesis.

3.2.3.3 Hypothesis 2: Progressive downstream migration of bed coarsening

Because reductions in sediment supply (in this case due to the dams) affect the reaches immediately downstream first, the bed coarsening process outlined above would have probably worked its way progressively downstream over time. As of 1980, the coarsening was thought to have affected the upper Sacramento River from Keswick Dam to at least as far downstream as RM 283 (i.e., Anderson Bridge) based on grain size analyses (CDWR 1980). The assumption that bed coarsening should propagate downstream as in-channel supply of coarse sediment from upstream reaches is exhausted leads to hypothesis 2: that coarsening has progressed downstream (to below RM 283) since 1980 and has now reduced spawning habitat area between Anderson Bridge and Cottonwood Creek.

3.2.3.4 Testing of hypothesis 2

There is little existing data to test this hypothesis. As noted above, an additional habitat survey, akin to the ones conducted in 1964 and 1980, is currently underway. It should help track the downstream propagation of bed coarsening. We expect that the local contributions of coarse sediment from small tributaries in the reach are unlikely to have masked the effects of coarsening, given that there have been several significant flow events in the mainstem Sacramento River since 1980. It should be possible to determine whether this is the case, based on analysis of the time series of habitat maps. The reach below Cottonwood Creek may have remained stable (in terms of spawning habitat) despite high flow events, due to decreased transport capacity (associated with local channel slope) or due to abundant sediment supply, which may have increased over the rates reported in Table 3-3, as gravel mining on Cottonwood Creek has been reduced since 1980. The habitat surveys should verify whether this is true. Results from TUGS simulations should provide additional information for assessment of hypothesis 2.

Large habitat losses from 1964 to 1980 might have slowed or stopped if remaining habitat is associated with relict features (such as point bars and riffles) that have local hydraulics which prevent high flows from eroding the remaining spawning gravel. We expect that habitat losses below Bend Bridge (RM 260) may have slowed or stopped since 1980, because bars and riffles have become relatively stable against scour. We therefore do not expect that the reach has coarsened significantly. This would have key implications for the population dynamics and management of fall-run Chinook salmon, which use the lower reaches (below RM 273.5) in addition to the upper Sacramento River (up to Keswick) for spawning. If the downstream propagation of bed coarsening has stalled at Cottonwood Creek, then adverse effects of reduced sediment supply for fall-run Chinook salmon may be limited to continued losses in RM 302–273.5. Conversely if bed coarsening continues to propagate downstream, then it is likely to have significant effects on fall-run Chinook salmon. Additional details about this and other concerns related to the specific runs of Chinook salmon are presented in Chapter 4.

3.2.3.5 Hypothesis 3: Increase in fine sediment in subsurface bed material

As noted in Section 3.1.3, bulk samples collected from RM 302–273.5 in 1995 indicate low rates of fine sediment infiltration and moderate to high gravel permeability. This runs counter to the general expectation that sand and finer material should accumulate in the bed in the absence of coarse material supply, due to the relatively high mobility of fine material, which is presumably supplied from tributaries, bank erosion, and agricultural runoff Whether this is the case below Cottonwood Creek has not yet been resolved. Increasingly high fine sediment concentrations in the bed below Cottonwood Creek would have important implications for management of fall-run Chinook salmon.

This leads to working hypothesis 3: fine sediment concentrations in the mainstem channel bed in the first few river miles below Cottonwood Creek are higher than they are above the confluence, due to the effects of dam-related reductions in peak flows, the relative high sediment supply (from the tributary). If present, high concentrations of fine sediment in the mainstem channel below Cottonwood Creek might be ameliorated by more frequent high flow events, which would tend to scour gravels and thus expose subsurface fine sediment to downstream transport.

It should be possible based on new field data on permeability and grain size (collected in the ongoing gravel study) to determine whether fine sediment accumulation below Cottonwood Creek has progressed to the point where it might have detrimental effects on fall-run Chinook salmon. Given that TUGS can be used to determine how the concentration of fine sediment evolves over time in the channel bed, it should help shed light on whether we should expect to see progressive increases in fine sediment in the reach below Cottonwood Creek, as proposed in hypothesis 3. In the gravel study, permeability data will also be collected from gravel bars in the upper river. This should make it possible to determine whether fine sediment deposition in the upper river is adversely affecting spawning gravel quality.

3.3 Meander Migration

In this section, we present a summary of the current understanding of meander migration on the Sacramento River. Meander migration affects the quality and availability of near- and off-channel habitat for a diverse array of species on the Sacramento River, including several focal species of this report (i.e., the bank swallow as discussed in Chapter 7, the western pond turtle as discussed in Chapter 8, and the Fremont cottonwood—a foundation species for riparian vegetation dynamics, as discussed in Chapter 9).

Meander migration and bank erosion occur by two processes: progressive channel migration (Figure 3-5A), in which flows erode banks incrementally, and episodic meander-bend cutoff, in which the channel avulses to a completely new course (Hooke 1984). Cutoffs may be partial (Figure 3-5B) or complete (Figure 3-5C), depending on initial meander bend geometry and the resistance of bank and floodplain materials to erosion, among other factors. Complete cutoffs are often referred to as "chute cutoffs". Partial cutoffs are sometimes also referred to as "neck cutoffs" in geomorphology texts and literature. While progressive migration and episodic cutoff can generally be thought of as distinct (i.e., mutually exclusive) processes, they are nevertheless interrelated, because they simultaneously regulate and are affected by sinuosity and other channel characteristics, as discussed in greater detail below.

Actively migrating reaches of the Sacramento River show evidence of repeated cutoff. Former channel alignments are preserved in the floodplain between RM 243 and RM 143 by long-lasting differences in topography, soils, and vegetation (e.g., Brice 1977; Micheli and Larsen, in preparation; Constantine et al. unpublished). Channel migration in the reach has historically been a key regulator of near- and off-channel habitat dynamics for riparian vegetation (Chapter 9) and many species, including the bank swallow (Chapter 7) and the western pond turtle (Chapter 8). To improve understanding of how habitat dynamics along the middle Sacramento River are affected by land- and water-use management, it is important to quantify rates and patterns of channel migration processes, and to determine how they are affected by changes in flow and land use.

Lateral shifts of river channels and changes in floodplain vegetation over time have often been quantified from temporal sequences of planform maps (e.g., Brice 1977, MacDonald et al. 1993, Gurnell et al. 1994, Brewer and Lewin 1998, Dietrich et al. 1999, Greco et al. 2003, Micheli et al. 2004). For the middle Sacramento River (RM 243–143), data on channel planform, soils, geology, and vegetative cover have now been assembled in a GIS format by CDWR, providing a ready means for accurately calculating rates of change from superimposed planform maps (e.g., Winterbottom and Gilvear 2000). This approach was used in two recent independent studies of a century-spanning time series of planform data (Micheli et al. 2004; Micheli and Larsen, in preparation; Constantine et al. unpublished). In both cases, the data was used to clarify the timing of individual cutoff events and produce a database of average rates of channel migration for the middle Sacramento. Results from these spatial analyses of planform changes provide the most comprehensive and quantitative data on rates and styles of meander migration on the Sacramento River. The broad spatial extent of the analysis (i.e., from RM 243 to RM 143) and long (i.e., decadal to centennial) timescales of the data make the results an ideal focus for the discussion presented here. Additional localized accounts of short-term bank erosion rates from ongoing observational studies (e.g., Buer 1994a) are provided as needed on a case-by-case basis in later discussions (e.g., particularly in Chapter 7), but are not summarized in exhaustive detail here. As discussed below, analysis of the Sacramento River planform data reveal that migration has varied in rate and style both over time and as a function of distance downstream (Micheli et al. 2004), showing especially sharp contrasts in rates of lateral change in an alternating series of stable and unstable reaches (Schumm and Harvey 1986, Constantine et al. unpublished).

3.3.1 Active and stable reaches of the middle Sacramento River

The reach between Red Bluff (RM 243) and Colusa (RM 143) has historically been more or less free to migrate, except where it is constrained locally by bridges and bank stabilization projects (Buer 1994a, Micheli et al. 2004). Several short stretches within the reach appear to have naturally slow rates of meander migration (Schumm and Harvey 1986). This is clearly evident when the overall rate of meander migration (i.e., due to progressive migration and cutoff combined) is plotted against river mile on the Sacramento River. A clear pattern of alternating "stable" and "active" reaches emerges (Schumm and Harvey 1986; Avery et al. 2003; Micheli and Larsen, in preparation; Figure 3-6), with differences in reach-averaged migration rates for adjacent reaches of up to an order of magnitude (Constantine et al. unpublished; Figure 3-7). Stable reaches are short (i.e., 3 river miles long or less), have persisted since long before bank protection projects began (i.e., for more than 100 years), and for the most part occur within areas that are underlain by terrace deposits (including the Tehama, Red Bluff, Modesto, and Riverbank formations), which locally confine the river between relatively resistant lateral boundaries (Figure 3-8), and thus appear to greatly reduce progressive migration rates and stall downstream migration of meander bends (Constantine et al. unpublished). Localized constraints on channel migration, such as banks where harder (less erodible) geologic deposits are exposed, may be particularly important for determining where and how off-channel habitats form. By creating relatively straight reaches with low channel mobility, erosionally resistant geologic deposits such as the Modesto and Riverbank formations can stall downstream migration of a meander train (Larsen and Greco 2002) and thus make migrating reaches immediately upstream especially predisposed to cutoff processes (Constantine et al. unpublished).

There is some indication that, within the seven active reaches that occur between RM 243 and RM 143, migration rates may be regulated, at least in part, by rates of sediment deposition on bars, with higher migration rates in bends that have higher deposition rates (Constantine et al. unpublished). This has been noted to be consistent with sediment transport theory (Lewin 1976, Dietrich and Smith 1983) and observations on other rivers (Dunne 1988, Ham and Church 2000) which indicate that local bed topography can help set lateral migration rates by focusing flow into the outside bend of the meander (Constantine et al. unpublished). Implications of bedform-related variations in migration rates are considered in the context of management issues (e.g., levee setbacks and riprap removal) in focal species chapters (e.g., in particular in Chapters 8 and 9) and the overall SOS Report synthesis (Chapter 10).

3.3.2 Relative importance of progressive migration and chute cutoff

Over the last roughly 100 years on the middle Sacramento River between RM 243 and RM 143, progressive migration has dominated over cutoff as a lateral migration mechanism, affecting roughly 93% of the total channel length on average versus just 7% for cutoff (Micheli and Larsen, in preparation, Table 3-4). In other words progressive migration has affected roughly 13 times more of the overall length of channel on average than cutoff has over an equivalent period of record. The share of the overall area eroded by migration, on the other hand, is disproportionately smaller for progressive migration, such that lateral shifts by progressive migration, at 0.28 mi^2 yr^{-1} (0.59 km^2 yr^{-1}), account for just 80% of the 0.29 mi^2 yr^{-1} (0.74 km^2 yr^{-1}) overall rate for all processes combined. This is because cutoff events, although relatively rare, tend to affect relatively large areas when they do happen (Micheli and Larsen, in preparation; Table 3-4).

Table 3-4. Relative importance of progressive migration and cutoff on the Sacramento River from RM 243 to 143.

Time interval	% of total channel length affected			Total floodplain area affected*					
	Progressive migration	Partial cutoff	Chute cutoff	Progressive migration		Partial cutoff		Chute cutoff	
				mi^2 yr^{-1}	km^2 yr^{-1}	mi^2 yr^{-1}	km^2 yr^{-1}	mi^2 yr^{-1}	km^2 yr^{-1}
1904–1937	92	1	7	0.192	0.499	0.003	0.009	0.047	0.121
1937–1952	91	2	7	0.271	0.703	0.007	0.018	0.064	0.167
Average pre–1952	91.7 ± 0.5	1.3 ± 0.5	7	0.22 ± 0.04	0.56 ± 0.10	0.04 ± 0.02	0.012 ± 0.005	0.05 ± 0.01	0.14 ± 0.02
1952–1964	97	1	2	0.196	0.508	0.006	0.016	0.033	0.085
1964–1978	92	2	6	0.243	0.629	0.012	0.032	0.065	0.168
1978–1987	94	2	4	0.343	0.888	0.023	0.060	0.067	0.173
1987–1997	96	1	3	0.184	0.477	0.006	0.016	0.037	0.095
Average post–1952	94.6 ± 1.1	1.5 ± 0.3	3.9 ± 0.9	0.24 ± 0.03	0.61 ± 0.09	0.010 ± 0.003	0.03 ± 0.01	0.05 ± 0.01	0.13 ± 0.02
Grand average	93.1 ± 1.0	1.4 ± 0.2	5.5 ± 0.9	0.23 ± 0.02	0.59 ± 0.06	0.007 ± 0.003	0.02 ± 0.01	0.05 ± 0.01	0.13 ± 0.02

* Cutoffs constitute discreet events which may often be bracketed (before and/or after) by progressive migration within any given interval. Limitations of the analysis, which relies on a time-series of planform data, render distinction of progressive migration from cutoff migration impossible for reaches that were affected by cutoffs (whether partial or complete). For the purposes of this analysis all of the eroded area in cutoff-affected reaches counts toward area affected by cutoff. This means that the area affected by partial and chute cutoff in any given interval is a maximum estimate. When cutoffs occur, they generally affect much larger areas (on a per-unit-length basis; see Table 3-5) than those affected by progressive migration. As a consequence, errors introduced by including area affected by progressive migration in the estimates of cutoff areas should be small enough to ignore in most cases.

Source: Micheli and Larsen, in preparation.

The relatively greater rate of cutoff migration (on a per-unit basis for centerline length) compared to progressive migration is illustrated in Table 3-4; on average, partial and chute cutoffs have produced 2–5 times more lateral channel change per unit stream length than progressive migration (Table 3-5). By providing rates of migration in individual intervals of time the migration analysis includes information about changes in channel migration rates over time. There is some indication from consideration of results in Tables 3-4 and 3-5 that the per-unit-length rate of migration by cutoffs (both partial and complete) has increased substantially in the post-dam era (i.e., represented here by photos postdating 1952)—by a factor of two for each type of cutoff migration. This apparent increase in average cutoff rates has been offset to a certain degree by a decreasing overall length of channel affected by cutoff processes over the same interval. Temporal variations in migration rates are discussed in greater detail in Section 3.2.6.

Table 3-5. Rates of progressive migration and cutoff on the Sacramento River from RM 243 to 143.

Time interval	Lateral migration rate						Average lateral migration rate[a]	
	Progressive migration		Partial cutoff		Chute cutoff		ft yr^{-1}	m yr^{-1}
	ft yr^{-1}	m yr^{-1}	ft yr^{-1}	m yr^{-1}	ft yr^{-1}	m yr^{-1}		
1904–1937	12.5	3.8	23.3	7.1	37.4	11.4	14.4	4.4
1937–1952	18.0	5.5	23.3	7.1	57.4	17.5	20.7	6.3
Average pre-1952	14.1 ± 2.6	4.3 ± 0.8	23.3	7.1	43.6 ± 10.2	13.3 ± 3.1	16.4 ± 3.3	5.0 ± 1.0
1952–1964	12.1	3.7	46.9	14.3	109.9	33.5	14.4	4.4
1964–1978	16.1	4.9	33.1	10.1	67.6	20.6	19.7	6.0
1978–1987	22.0	6.7	83.3	25.4	47.1	29.6	26.2	8.0
1987–1997	11.5	3.5	45.9	14.0	65.6	20.0	13.8	4.2
Average post-1952	15.1 ± 2.3	4.6 ± 0.7	49.5 ± 10.8	15.1 ± 3.3	84.3 ± 10.8	25.7 ± 3.3	18.4 ± 3.0	5.6 ± 0.9
Grand average	14.8 ± 1.6	4.5 ± 0.5	36.1 ± 9.2	11.0 ± 2.8	63.3 ± 10.8	19.3 ± 3.3	17.4 ± 2.0	5.3 ± 0.6

[a] due to progressive, partial cutoff, and chute cutoff combined.
Source: Micheli and Larsen, in preparation.

3.3.3 Progressive meander migration

The progressive migration of a meander bend occurs via the gradual erosion of the outside (i.e., concave) bank and deposition along the inside bank on the point bar. Bank erosion on an outside bank is generally balanced by point bar deposition on the corresponding inside bank such that channel width remains roughly constant as the river shifts both laterally and in the downstream direction on its floodplain (Lawler 1993).

3.3.3.1 Controls on progressive meander migration on the Sacramento River

As a general rule, rates and patterns of progressive migration reflect the balance between shear forces of flow at the bank and the resistance of bank and floodplain materials to erosion (Ikeda et al 1981, Howard and Knutson 1984). Compared to inside bends, outside bends typically have greater depth, velocities, and higher shear forces acting along the toe of the bank. These sites typically exhibit relatively rapid progressive migration rates (Thorne 1992). Channel curvature is another important regulator of shear forces acting on channel banks (Johanesson and Parker 1989), with higher curvature corresponding to locally higher rates of bank erosion (Nanson and Hickin 1986, Furbish 1988).

In general, alluvial bank materials are expected to be increasingly cohesive (and therefore less erodible, and thus more capable of forming sinuous meander bends) as floodplain deposits become increasingly finer (Knighton 1984). Native riparian vegetation also increases the effective cohesion of bank and floodplain soils and the hydraulic roughness of the channel (Thorne 1992). Bank resistance to erosion (the inverse of "bank erodibility", as defined by many meander migration models) has been observed to vary with floodplain geology and riparian vegetation on the Sacramento River (Larsen and Greco 2002, Micheli et al. 2004, Constantine et al. unpublished). As discussed at greater length in Chapters 7 and 9, removal of riparian vegetation on the Sacramento River has been correlated with a doubling of channel migration rates (Micheli et al. 2004).

On the Sacramento River, bank erosion generally occurs in a two-step process (Buer 1984, Buer 1994a; K. Buer, pers. comm., 2000):

1. gradual erosion of the concave bank via fluvial entrainment of non-cohesive sediment at the bank toe
2. slab failure of overlying cohesive floodplain deposits.

The timing and magnitude of bank erosion can be affected by bank saturation, with slab failures tending to occur on the falling limb of the hydrograph (and presumably during other periods of rapid flow fluctuation), when positive pore pressures reduce the stability of bank soils against erosion (Thorne 1992). Recessional limb failures such as these have been observed in bank erosion studies of the Sacramento River (e.g., Buer 1994a). The effect of such failures on the overall rate of migration rate is difficult to quantify in the absence of data. If recessional limb failures are important on the Sacramento River, migration rates may be correlated to some extent with management of flow fluctuations via reservoir operations. A failure that apparently affected bank swallow habitat and may have been related to flow management is discussed in Chapter 7.

3.3.3.2 Modeling

Principles of fluid mechanics and sediment transport theory can be used to model the evolution of channel planform over time (Ikeda et al. 1981). In one common approach, a given cross sectional geometry is assigned a "representative" or "formative" flow (Larsen 1995). Planform curvature helps determine shear stresses at the outside bank. The model requires calibration to determine bank erodibility, which is generally expected to vary along the channel centerline (Hasegawa 1989, Pizzuto and Melckenburg 1989, Larsen and Greco 2002). Model outputs include linear cross-sectional profiles of bed elevation and depth-averaged flow velocity (Johanneson and Parker 1989, Larsen 1995). Once calibrated, the model can be used to predict progressive rates of channel migration for different management scenarios (e.g., alteration of the flow regime, removal of riprap, setback of levees), and can be customized to predict where new cutoffs are most likely to occur (Larsen et al. 2006).

3.3.3.3 Observations

Progressive meander migration rates averaged over a wide range of scales on the middle Sacramento River (from RM 243 to RM 143) are 8.2–16.4 ft y^{-1} (2.5–5.0 m y^{-1}), which is on the order of 0.01 to 0.02 channel widths per year (Table 3-6), a typical range for big rivers (Dietrich et al 1999, Larsen 1995). Individual bends on the Sacramento River have been observed to migrate up to ten times faster, at up to 0.10 channel widths per year or more (Harvey 1989). Studies of meander dynamics of the Sacramento River suggest that a radius of curvature (R/w)[3] of approximately 2.5 times the channel width corresponds with peak rates of channel migration (Schumm and Harvey 1986), consistent with findings of previous studies on other rivers (Hickin and Nanson 1984). Bends with radii of curvature over 1,250 ft (380 m) appear to be eroding the fastest on the middle Sacramento River (Harvey 1989). Relatively high meander migration rates appear to be concentrated downstream of a series of major Sacramento River tributaries, from RM 200 to RM 165 (Figure 3-7), in a zone that has been noted for its relatively high rates of sediment deposition (Constantine et al. unpublished). Progressive migration rates have remained roughly stable over time, within estimated uncertainties; the overall average rate of progressive migration along the middle Sacramento River in the post-dam interval (after 1952) was 15.1 ± 2.3 versus 14.1 ± 2.6 ft yr^{-1} (4.6 ± 0.7 versus 4.3 ± 0.8 m yr^{-1}) in the pre-dam interval (Table 3-6). This pattern persists at the local scale; reach averaged rates of progressive migration in the pre- and post-dam intervals agree within estimated uncertainties, with few exceptions (Figure 3-7).

Table 3-6. Empirical measurements of bank erosion rates.

Author, year	Time interval	Geographic scope	Meander migration rate		Meander migration rate (widths per yr*)
			ft yr^{-1}	m yr^{-1}	
US Congress 1960†	1896–1946	Chico Landing to Colusa	16.1	4.9	0.02
Brice 1977	1896–1948	Chico Landing to Colusa	17.4	5.3	0.02
Brice 1977	1896–1974	Chico Landing to Colusa	15.1	4.6	0.02
Buer 1994a	1986–2001	12 cross sections	8.2	2.5	0.01
Harvey 1989	1896–1986	Glenn to Chico, sinuous bends	82	25	0.10
Micheli et al. 2004	1896–1946	Red Bluff to Colusa	9.2	2.8	0.01
Micheli et al. 2004	1946–1997	Red Bluff to Colusa	13.8	4.2	0.02
Micheli and Larsen, in prep.	1904–1997	Red Bluff to Colusa	15.4	4.7	0.02

---

[3] Note that "radius of curvature" is not the same as "curvature". They are, in fact, inversely correlated: as radius of curvature increases, curvature (and sinuosity) decreases. For the extreme case of a straight line, curvature is zero and radius of curvature is infinite. Conversely for vanishingly small circle, curvature goes to infinity as radius goes to zero.

Author, year	Time interval	Geographic scope	Meander migration rate		Meander migration rate (widths per yr*)
			ft yr^{-1}	m yr^{-1}	
Constantine et al. unpublished	1896–1997	Active reaches RB to Colusa	9.5–19.7	2.9–6.0	0.01–0.02
Constantine et al. unpublished	1896–1997	Stable reaches RB to Colusa	0.3–6.9	0.1–2.1	< 0.01
Larsen et al. *in press*	1937–1975	Pine Creek Bend	6.6	~2	0.01
Buer 1994a	1986–2001	12 cross sections	8.2	2.5	0.01

* average channel width of 820 ft (250 m) is assumed
† as cited in Brice 1977.

3.3.3.4 Thresholds of bank erosion

From a habitat management perspective, it is important to understand how progressive migration rates are likely to vary with discharge over time. As discussed at greater length in Chapter 7, a particular concern for management of bank swallow habitat is the local flow threshold for bank erosion. Progressive migration can occur over a range of flows, with higher rates generally associated with higher flows. Bank erosion rates are generally assumed to peak during periods of bankfull flow, when flow shear at the toe is maximized (Johannesson and Parker 1989, Leopold 1994).

Fluctuations in stream flow over time will lead to fluctuations in bank erosion rates (Hooke 1980). For a series of bends on the middle Sacramento River, short-term bank erosion rates (averaged over the interval 1981–1986) exceed long-term rates (averaged over the interval 1896–1986) by 57–73% (Harvey 1989). This discrepancy has been attributed to a prevalence of relatively wet conditions during the shorter interval (Harvey 1989); on average, the wetter years presumably contributed more flows above the bank erosion threshold and thus contributed more energy for progressive lateral migration. This is consistent with the idea that cumulative bank erosion in any given interval should be related to the cumulative duration of flows above the bank erosion threshold.

The indication that bank erosion is tied to cumulative flow above a certain threshold has been supported by other studies of the middle Sacramento River. Analysis of nearly a decade's worth of data from cross-sectional surveys (Buer 1994a) shows that variations in annual bank erosion from sites between RM 194 and RM 165 in the post-dam interval are tightly coupled with variations in cumulative annual stream power (Larsen et al., unpublished). The correlation, based on data from relatively dry years, is shown in Figure 3-9 for two threshold discharges: 15,000 cfs and 29,500 cfs. A relatively better correlation is observed for a threshold discharge of 29,500 cfs.

Field observations have variously indicated thresholds for bank erosion on the Sacramento River ranging from as low as 7,500 cfs for localized erosion to 60,000 cfs for widespread bank erosion (Table 3-6; Kondolf et al. 2000, K. Buer, pers. comm., 2005). These estimates are all well below the range of available estimates of bankfull flow (Table 3-6), which spans 88,300–141,300 cfs. In general, for the river as a whole, the balance of evidence suggests that extensive episodes of widespread bank erosion can occur during moderate to major flow events. The wide scatter in bank erosion thresholds in Table 3-5 suggests that local differences in channel geometry and bank materials may lead to significant reach-to-reach variability in the threshold of bank erosion. Examples of localized bank erosion are discussed in Chapter 7.

Table 3-7. Bank erosion thresholds and several "formative" discharges as estimated by various sources.

Threshold condition (and basis for estimate)	Author and year	Discharge (cfs)
Bank erosion threshold (analysis of cumulative effective discharges at 7 sites)	Kondolf et al. 2000	7,500–29,500*
Bank erosion threshold (field observations)	Larsen et al. 2006	10,600
Bank erosion threshold (field observations)	Buer 1995, Klinesteker 1998	> 13,000
Bank erosion threshold for 3 sites (based on threshold bank erosion velocities reported by US Congress 1960†)	Brice 1977	22,000–32,000*
Widespread bank erosion (field observations)	K. Buer, pers. comm., 2005 (also as cited in Larsen et al., unpublished)	60,000
1.5-yr flood (calculated from hydrograph)	Kondolf et al. 2000	61,000 (86,000**)
Bankfull discharge (estimated)	Thomas 2000	88,300 (141,300**)
2-yr flood (calculated from hydrograph, Vina gauge, 1964–1980)	Larsen and Greco 2002 (their "characteristic discharge")	96,100

* lower and upper bounds on flows are indicated if available
** higher number corresponds to pre-dam conditions
† as cited in Brice 1977.

3.3.4 Dynamics of cutoff and off-channel habitat formation

Rates and patterns of progressive migration control the generation of bends and regulate their geometry, which in turn influences their susceptibility to cutoff. As discussed in the preceding section, rates of progressive migration are thought to generally increase with curvature. But this may be true only up to a point. At high curvatures, above a certain threshold, bends can become so tight that they generate backwater effects which reduce the energy available for bank erosion (Hickin and Nanson 1984, Furbish 1988). In bends such as these, increases in flow may increase water depths enough to initiate overbank flow, thus leading to erosion on the floodplain and potentially initiating chute cutoff—a rapid shift in channel alignment due to sediment scour on the floodplain.

Cuttoffs generate side channels, which are eventually converted to sloughs and oxbow lakes, which eventually fill in through a process referred to by some as "terrestrialization" (e.g., Piégay et al. 2000, Morken and Kondolf 2003). Although cutoffs on average affect only 7% of the migrating section of the Sacramento River, they are extremely important because they provide the only mechanism of off-channel habitat creation. Cutoffs also provide an important pathway for the establishment of Fremont cottonwood and other riparian plant species (see Chapter 9).

3.3.4.1 Modeling cutoff formation

The majority of river migration models focus on dynamics of single thread channels, because the full complexity of natural channels is difficult to express mathematically (Ikeda et al. 1981, Howard and Knutson 1984, Johannesson and Parker 1989). As a result, models of cutoff formation are not as well developed as progressive migration models; no model has been able to comprehensively predict rates of off-channel habitat creation and in-filling. Conventional understanding of how off-channel habitat is affected by cutoff formation has therefore been largely derived from field observations rather than modeling exercises. For the Sacramento River, particularly enlightening results have been compiled from interpretations of the relative importance of progressive and cutoff migration (Micheli et al. 2004; Constantine et al. unpublished*;* Micheli and Larsen, in preparation) and data collected in a long series of empirical studies (e.g., Brice 1977, Buer et al. 1989, WET 1990, USACE 1991, Buer 1994a, Larsen and Greco 2002, Greco and Plant 2003, Morken and Kondolf 2003).

3.3.4.2 Cutoff formation processes

Channel cutoffs generally occur via the following sequence of processes:

1. Over-bank flows cause a "probe" channel—a precursor to the chute—to be scoured across the floodplain.
2. The probe develops to the point where it connects an upstream point of a sinuous bend with a point further downstream and thus provides a shortcut for the mainstem flow.
3. If overbank flow is deep enough and persists for long enough, the probe may expand into a complete cutoff (capturing all of the river's flow). The expansion is generally thought to progress by upstream migration of a knickpoint, which is typically initiated by oversteepening and mass failure at a plunge-pool were the probe initially rejoins the mainstem (Gay et al. 1998).

Based on an assumed typical bank height of 26 ft (8 m) and an assumed typical channel width of 820 ft (250 m), the formation of a typical chute of 3,281 ft (1000 m) length entails excavation of approximately 2.6 million ft^3 (0.2 million m^3) of floodplain material (Micheli and Larsen, in preparation). Chutes of this scale have been observed to form on the Sacramento River within a single winter season (Micheli and Larsen, in preparation).

Probes that span the entire bend from one (upstream) inflection point to the next (downstream) become complete chute cutoffs. Probes that cross only a portion of the bend will become partial cutoffs (Figure 3-5). On the Sacramento River over the last roughly 100 years, chute cutoffs have been more common than partial cutoffs (Buer 1994a, WET 1990), outnumbering them by slightly more than three to one (Table 3-4; Micheli and Larsen, in preparation).

3.3.4.3 Formation and terrestrialization of sloughs and oxbow lakes

After the bulk of the river's flow occupies the new main channel, the cutoff meander bends are relegated to "side channel" status. These side channels eventually become plugged with coarse sediment, first at the upstream end (creating an oxbow slough) and then later sometimes at the downstream end as well (creating an oxbow lake). Infilling of oxbow sloughs and lakes is generally gradual with addition of fine sediments that "terrestrialize" off-channel water bodies arriving in infrequent overbank floods (Morken and Kondolf 2003). Sloughs and oxbow lakes can provide refuge for aquatic species. Sloughs maintain a roughly constant connection to main channel's flow, whereas oxbows become connected to the main channel intermittently, during relatively high flows, when stage is sufficient to run over the top of the plugs. In some cases, this may occur at flows as low as 10,000–15,000 cfs (G. Golet, pers. comm., 2006), whereas in others, full, overbank flows may be required. Determining how and when individual oxbows become connected to the mainstem is crucial for understanding the evolution and relative importance of each of the various off-channel habitats on the Sacramento River floodplain.

Rates of terrestrialization have only been quantified in a few instances, but are generally expected to be highly variable, due to high spatial variability in frequency of overbank flow and fine sediment deposition rates. In one field study it was shown that that off-channel habitats on the Sacramento River can retain aquatic habitat value—i.e., with open water surrounded by early-succession riparian forest and wetlands—for over 50 years (Morken and Kondolf 2003). Small channels connecting the downstream ends of sloughs to the main channel have been observed to remain open for up to 15 years (Morken and Kondolf 2003). Significantly older connections are probably common, although difficult to confirm for Sacramento River sloughs in the absence of observational data.

3.3.5 Geometry of migrating meander bends

Analysis of planform geometry indicates there are systematic differences in bends that have been affected by different types of migration. For example, meander bends that have migrated via progressive migration are typically much less sinuous than bends that migrate laterally via cutoff (Micheli and Larsen, in preparation, Table 3-8). From a mechanistic standpoint, this is not altogether surprising; all else equal, the more sinuous the bend, the shorter should be its required chute cutoff length, implying a correspondingly higher likelihood of cutoff formation.

Table 3-8. Planform geometry of migrating bends on the Sacramento River, 1904–1997.

Type of migration	Total number of bends	Sinuosity (M/L)*	Average dimensionless radius of curvature (R/w)†	Mean entrance angle (degrees)
Chute cutoffs	27	1.97 ± 0.1	2.1 ± 0.2	111 ± 7
Partial cutoffs	11	1.43 ± 0.1	2.3 ± 0.3	77 ± 9
Stable high-sinuosity (> 1.85)	35	2.24 ± 0.1	2.5 ± 0.1	112 ± 4
Progressive migration	328	1.31 ± 0.01	2.8 ± 0.1	66 ± 1
Oxbow Lakes (1997 only)	NA	2.01 ± 0.17	2.2 ± 0.2	NA

* Defined as the meander bend length (M) divided by local valley length (L) (after Larsen et al. unpublished).
† Defined as the radius of curvature (R) (see footnote 2) divided by channel width (w).
Source: Micheli and Larsen, in preparation.

Entrance angle helps determine the degree to which the flow is focused on (and thus exerts shear stress on) the upstream inflection point. Higher angles are associated with higher erosive forces on the bank and higher likelihoods of probe channel initiation. This is consistent with the fact that entrance angles are higher in bends that have been affected by chute cutoff (Table 3-8).

Radius of curvature is thought to be another important regulator of lateral channel changes over time, particularly in bends that are affected by progressive migration (which is influenced by shear stress at the bank). A dimensionless radius of curvature (i.e., normalized by channel width) is generally used for ease of comparison among rivers which may vary greatly in scale (Hickin and Nanson 1984, Harvey 1989). There is some indication that cutoff processes may have a characteristic threshold value of dimensionless radius of curvature that varies from river to river (Hickin and Nanson 1984, Harvey 1989). For the Sacramento River, the average radius of curvature of bends ranges from 2.1 for bends that have been affected by chute cutoff to 2.8 for bends that have been affected by progressive migration (Table 3-8). This is consistent with the expectation that curvature in a progressively migrating bend will tend to increase, but only up to a point before cutoff reduces curvature by providing a short cut for the flow. Radii of curvature shown in Table 3-8 are consistent with a threshold for cutoff > 2.8.

3.3.6 Geomorphic metrics of meander migration

The interrelationship between chute cutoff and progressive migration should produce a characteristic pattern of planform evolution over time. Individual bends should tend to evolve greater sinuosity and curvature via progressive migration channel. When the local cutoff threshold is reached, an avulsion may occur and reduce the overall sinuosity of the affected reach. This should lead to measurable changes in local geomorphology over time. Assuming that reductions (or increases) in metrics such as sinuosity in one reach are balanced by increases (or reductions) elsewhere, the overall pattern of planform geometry for the migrating reaches of the Sacramento River as a whole might be expected to approach a state of dynamic equilibrium. Given that lateral migration processes are the key regulators of near- and off-channel habitat on the Sacramento River, it seems likely that under dynamic equilibrium, the formation and destruction of key focal species habitat features would be balanced by lateral migration processes. If lateral migration of the Sacramento River is not in a state of rough dynamic equilibrium, then we expect to see trends in planform geometry and/or migration rates over time. We explore whether this has been the case in the sections below.

3.3.6.1 Trends in geometry of cutoff bends

Over the last roughly 50 years, average sinuosity has been lower and the average radius of curvature of cutoff bends has been higher relative to what they were in the early 1900s before the dams were built and when the Sacramento River floodplain was being rapidly converted to agricultural uses. Cutoff bend sinuosity has steadily declined 30% in a series of six time steps (from 2.3±0.4 in 1904–1937 to 1.5±0.2 in 1987–1997) (Table 3-9; Figure 3-10). Mean values for cutoff bend radius of curvature have increased 50% from a low of 1.6 in 1952–1964 to a high of 2.4 in 1987–1997 (Table 3-9). These trends suggest that it may be getting progressively easier for the river to excavate chute cutoffs across the floodplain, even as the energy available to create cutoffs has presumably decreased due to dam-related reductions in the magnitude and duration of overbank floods. This could be explained, at least in part, by increases in erodibility and reductions in roughness associated with the expansion of agriculture and other land uses which resulted in progressive removal of riparian forest vegetation throughout much of the 20th century (Micheli and Larsen, in preparation).

Table 3-9. Planform geometry of cutoff bends on the Sacramento River, 1904–1997.

Time interval	Number of bends	Mean sinuosity	Dimensionless mean radius of curvature (R/w)	Mean entrance angle (degrees)
1904–1937	6	2.3 ± 0.4	2.3 ± 0.4	117 ± 5
1937–1952	6	2.1 ± 0.3	1.9 ± 0.3	124 ± 18
1952–1964	2	2.0 ± 0.2	1.6 ± 1.0	71 ± 9
1964–1978	6	1.8 ± 0.2	2.0 ± 0.4	113 ± 17
1978–1987	4	1.8 ± 0.2	2.3 ± 0.8	110 ± 24
1987–1997	3	1.5 ± 0.2	2.4 ± 0.5	101 ± 20
Average		2.0 ± 0.1	2.1 ± 0.2	111 ± 7

3.3.6.2 Trends in frequency and timing of cutoff bends

There is some indication, from the available planform record of the Sacramento River (Micheli and Larsen, in preparation), that the ratio of occurrence of chute cutoffs to partial cutoffs may be decreasing over time. As noted above the overall ratio over the 93 year period of record is just over 2.4:1. However, before 1952, the ratio was 3:1 (i.e., 12 chute cutoffs to 4 partial cutoffs), compared to just over 2:1 (15:7) after 1952 in the post-dam interval (Table 3-9; Micheli and Larsen, in preparation).

A plausible explanation for the relative increase in frequency of partial cutoffs is the progressive decline in riparian forest cover over time. Riparian forest vegetation appears to reduce the erodibility of river banks and floodplains, such that clearing it from the floodplain may help decrease the threshold sinuosity for meander bend cutoff. If this is the case then it implies that agriculturally motivated changes in vegetative cover may result in the development of relatively straight meander bends over time. Over the long term, this would tend to produce relatively straight oxbow lakes and sloughs, and it could also exacerbate downstream flooding.

In addition to affecting cutoff shape the increased frequency of partial cutoffs implies that average cutoff size may be changing over time as well. This point is confirmed by analysis. Average area affected by cutoff in the pre-dam interval, at 0.20 ± 0.03 mi^2 (0.52 ± 0.07 km^2), was slightly higher than the 0.13 ± 0.03 mi^2 (0.34 ± 0.09 km^2) post-dam average (Table 3-10). This indicates that cutoff size has decreased over time as the frequency of cutoffs has increased. It is further confirmed by inspection of Figure 3-11 which shows the location, magnitude, and timing of the 38 cutoffs that occurred over the period 1904–1997.

Table 3-10. Cutoff frequency and area, middle Sacramento River.

Time interval	Number of cutoffs	Number partial cutoffs	Number chute cutoffs	Average number of cutoffs per year	Average cutoff area	
					mi^2	km^2
1904–1937	7	1	6	0.21	0.24 ± 0.04	0.61 ± 0.10
1937–1952	9	3	6	0.60	0.12 ± 0.03	0.31 ± 0.08
pre-1952	*16*	*4*	*12*	*0.33*	*0.20 ±0.03*	*0.52 ±0.07*
1952–1964	3	1	2	0.25	0.15 ± 0.11	0.40 ± 0.29
1964–1978	9	3	6	0.64	0.12 ± 0.05	0.31 ± 0.12
1978–1987	6	2	4	0.67	0.14 ± 0.02	0.35 ± 0.06
1987–1997	4	1	3	0.40	0.11 ± 0.02	0.28 ± 0.06
post-1952	*22*	*7*	*15*	*0.49*	*0.13 ±0.03*	*0.34 ±0.09*
Grand totals and averages	*38*	*11*	*27*	*0.41*	*0.17 ±0.02*	*0.43 ±0.06*

*Reach sinuosity measured as total stream length divided by total valley axis length for the initial channel centerline.
Uncertainties are standard errors of the mean.

On average, a cutoff event has occurred on the middle Sacramento River roughly every other year (0.41 per year). The historical record shows that cutoffs are clustered during especially wet winters, with flooding in the winters of 1969–1970, 1974–1975, and 1983–1984 each producing multiple cutoff events (Micheli and Larsen, in preparation). Antecedent conditions may be an important factor as well. For example, high soil moisture in migrating knickpoints of probe channels may help regulate cutoff frequency, if they affect bank stability, and thus the likelihood of continued knickpoint migration. If they do, then rapid flow fluctuations may promote cutoff in some cases by increasing pore pressures (and thus water contents) of bank materials in probe channel knickpoints. Cutoff frequency increased from 0.33 cutoffs per year in the pre-dam era to 0.48 cutoffs per year in the post-dam era (Table 3-11).

Table 3-11. Cutoff frequency and overbank flow, Sacramento River, Bend Bridge Gauge.

Time interval	Total number of cutoffs	Average number of cutoffs per year	Peak discharge (cfs)	Number of days $Q_{1.5}$ is exceeded	Average annual overbank discharge	
					$10^8\ ft^3$	$10^6\ m^3$
1904–1937	7	0.21	252,005	41	3.16	89.5
1937–1952	9	0.60	290,993	17	3.50	99.2
pre–1952	*16*	*0.33*	*290,993*	*58*	*3.25*	*92.1*
1952–1964	3	0.25	138,999	22	1.82	51.5
1964–1978	9	0.64	157,009	33	4.09	115.7
1978–1987	6	0.67	151,994	36	0.04	114.3
1987–1997	4	0.40	126,992	18	2.43	68.9
post–1952	*22*	*0.49*	*157,009*	*109*	*3.10*	*87.9*

Once a bend has become sinuous enough to be susceptible to cutoff, the timing of the actual avulsion will be driven by the occurrence of overbank flows. As a working hypothesis we assume that the initiation of cutoff processes is tied to integrated magnitude and duration of overbank flow (Micheli and Larsen, in preparation). The cumulative volume of overbank flow estimated for the post-dam time intervals of the planform analysis is correlated with cutoff frequency (Figure 3-12, Table 3-11). The cumulative overbank flow of the pre-dam interval was characterized by a low cutoff frequency, at least compared to what one would predict from the trend exhibited by post-dam data (Figure 3-12). This may be a further indication of a fundamental change in the erodibility of the floodplain due, for example, to progressive removal of riparian vegetation over the last century. Alternatively it could reflect a key limitation of the data, namely that the number of cutoffs may have been underestimated for the pre-dam interval if (as may be reasonably expected) the older basemaps have lower resolution than those that pertain to later intervals.

A third possibility is that the discrepancy reflects the effects of bank stabilization projects. Riprap installation was progressive throughout the mid to late 20th-century, such that its effects were coincident with the effects of the dams and vegetation removal. Teasing out the relative importance of each potential factor is problematic.

In general quantifying the effects of riprap on trends in migration rates and processes is difficult. In the analysis of Constantine et al. (unpublished), migration rates for each eroding bank were averaged over only the period that the river was locally free to migrate—in other words, if a section became riprapped during an analysis interval, the migration rate was calculated by dividing the eroded area by the time between the start of the interval and the date of bank stabilization (Constantine et al. unpublished). However, although Constantine et al. were able to account for riprap, they did not explicitly consider its effects on the relative importance of progressive and cutoff migration processes. Conversely, whereas Micheli and Larsen (in preparation) did seek to distinguish progressive from cutoff migration, they did not account for effects of riprap at all. This makes any assessment of the effects of riprap on progressive migration and cutoff processes somewhat speculative based on available data and analyses from the Sacramento River. In general, we expect that riprap should substantially reduce progressive migration rates and should discourage probe channel formation (a necessary precursor to cutoff processes). To the extent that it does, it may contribute to some of the trends discussed in this section. In particular, a reduction in cutoff bend sinuosity (Table 3-9) and a decrease in average cutoff area (Table 3-10) may be attributable at least in part to effects of bank stabilization. Riprap may also locally affect a river's cross-sectional geometry, as discussed below.

3.3.6.3 Changes in cross-sectional geometry near riprapped banks

As part of their bank erosion monitoring program, CDWR measured thalweg depths and mean channel widths associated with 30 eroding banks and 37 riprapped banks between Red Bluff and Ord Ferry (Buer 1994a). Results showed that thalwegs along riprapped banks were an average of 6 feet deeper than thalwegs along eroding banks. Mean channel widths were an average of 70 feet narrower in the riprapped sections. A repeat analysis, four years later, confirmed the earlier results. Observations of relatively deep thalwegs and narrow widths in reaches that have been stabilized are consistent with the hypothesis that rivers that can no longer erode their banks will tend instead to erode their beds (Buer 1984). If this is the case then riprapping may locally reduce hydraulic diversity and salmon spawning area (Buer 1994a). Relatively narrow channels with deep thalwegs and steep cross-sectional geometries may also affect riparian succession, by supporting point bars that are too steep for efficient seed dispersal (see Chapter 9 for further discussion on riparian succession). On the other hand, an increase in the number and extent of deep pools may have increased available holding habitat for green sturgeon, as discussed in Chapter 6.

3.4 Synthesis of Geomorphic Controls on Focal Species Habitat

Suitable spawning habitat on the mainstem currently extends from Keswick Dam at RM 302 to Princeton at RM 163. Since 1945, Shasta (and later Keswick) Dam has altered mainstem flow and sediment supply, and has thus affected the quantity and grain-size distributions of gravel in the channel bed. This in turn has affected the extent and quality of salmonid spawning habitat. The expected evolution of spawning gravel in the Sacramento River can be summarized in the following three working hypothesis:

1. Bed coarsening in the upper Sacramento River has occurred and is continuing such that spawning habitat has been progressively reduced in the reach between Keswick Dam (RM 302) and Anderson Bridge (RM 283), despite the effects of recent gravel augmentation.
2. Bed coarsening has progressed downstream since 1980 and has now reduced spawning habitat area between Anderson Bridge (RM 283) and Cottonwood Creek (RM 273.5).
3. Although the concentration of fine sediment in the subsurface has appeared to remain suitably low between Keswick Dam (RM 302) and Cottonwood Creek (RM 273.5), it may have become higher in downstream reaches, due to dam-related reductions in peak flows coupled with high sediment supply from Cottonwood Creek (RM 273.5) and local hydraulic conditions (i.e., a break in slope) that promote local deposition, such that successful spawning of fall-run Chinook salmon in reaches below Cottonwood Creek (RM 273.5) may have been compromised.

These hypotheses are currently being tested as part of the Sacramento River Ecological Flows Study. Additional considerations of gravel dynamics as they relate to salmonid spawning and the various life stages of Chinook salmon, *Oncorhynchus mykiss,* and green sturgeon are presented in Chapters 4–6.

Whereas success of anadromous salmonids depends strongly on gravel dynamics in the mainstem, the other focal species considered in this report (Chapters 7–9) rely much more heavily on the dynamics of meander migration, which affects the quality and availability of near- and off-channel habitat. On the Sacramento River, actively migrating reaches alternate with stable reaches, which migrate slowly or not at all because they are confined by erosion-resistant geologic deposits or revetment placed to protect adjacent uses. Meander migration and bank erosion occur by progressive channel migration and episodic meander-bend cutoff. Over decadal timescales cutoffs generally affect less than 10% of the actively migrating length of the Sacramento River. Even so, cutoffs can account for well over 20% of the integrated lateral channel change, because they affect relatively large areas when they do occur.

Chute cutoff and progressive migration interact to produce a characteristic pattern of planform evolution over time. Individual bends evolve greater sinuosity and curvature via progressive channel migration. Cutoffs reduce sinuosity when it exceeds a local threshold for the initiation of cutoff processes. This should produce measurable changes in local geomorphology over time. Averaged over larger scales, however, changes in morphology in one reach should be balanced by changes in morphology in others, such that the overall pattern of planform geometry for migrating portions of rivers should approach a state of dynamic equilibrium in the absence of human modifications. Results presented here indicate that the sinuosity of cutoff bends is decreasing over time on the Sacramento River. This suggests that the Sacramento River is not in a state of dynamic equilibrium. The fact that cutoff migration has increased in frequency and is becoming increasingly dominated by partial cutoffs (which affect smaller areas compared to complete cutoffs) provides further evidence that non-equilibrium conditions may prevail.

Process-based considerations suggest that dam-related changes in flow (i.e., reductions in peak flow and cumulative over-bank discharge) should tend to reduce the frequency of channel cutoffs. This would generally be complemented by a reduction in average sinuosity, which appears to be underway based on available data (presented above). But observations from the Sacramento River indicate that the overall number of channel cutoffs has nevertheless increased in recent times. This supports the hypothesis that the erodibility of banks and floodplains has increased (and thus enhanced the likelihood of cutoff) due to the effects of agricultural clearing of riparian forests on floodplains (Micheli et al. 2004).

From a management perspective it is important to recognize the interrelated nature of progressive migration and meander bend cutoff. If progressive migration rates are slow, as is likely to be the case when significant stretches of bank are riprapped, then bends are unlikely to develop the high curvature necessary for chute cutoff and the formation of off-channel habitats. Cutoffs that do occur are likely to be relatively straight and short. This is consistent with the observed increase in importance of partial cutoffs relative to complete cutoffs over the last 100 years, as an increasingly larger percentage of the total bank length on the Sacramento River has been stabilized against lateral migration.

The following questions are being considered with the help of field data collection, analysis, and modeling in the Sacramento River Ecological Flows Study:

- Has the bed in the upper Sacramento River coarsened over time?
- Has the hypothesized coarsening propagated downstream over time?
- Is the accumulation of fine sediment in the channel bed a limiting factor for salmonid spawning habitat?
- How has flow regulation affected overbank floods?
- What are the implications of flow regulation for off-channel habitat formation?
- If the Sacramento River is becoming progressively straighter due to effects of human alterations to the ecosystem, will newly generated oxbow lakes and sloughs, being likewise straighter, have adequate complexity for ecological health. (Specific indicators of health are considered in later chapters.)
- What can be learned about thresholds for meander bend cutoff by studying individual cutoff events as they happen?
- What do newly available Geographic Information System (GIS) analyses of planform change indicate about spatial and temporal variations in extent of off-channel habitats?
- How does this relate to what we know about variations in lateral migration rates?
- Will reduced floodplain deposition rates extend the life of current and future off-channel habitats
- If so, what consequences will this have for the ecosystem?

These questions will be considered in greater depth in the context of the focal species analyses that follow in Chapters 4–9.

3.5 References

Avery, E. R., E. R. Micheli, and E. W. Larsen. 2003. River channel cut-off dynamics, Sacramento River, California, USA. EOS, Transactions of the American Geophysicists Union, 84(46), Fall Meeting Supplement, Abstract H52A-1181.

Brewer, P. A., and J. Lewin. 1998. Planform cyclicity in an unstable reach: complex fluvial response to environmental change. Earth Surface Processes and Landforms 23: 989–1008.

Brice, J. 1977. Lateral migration of the middle Sacramento River, California. Water-Resources Investigations 77–43. U.S. Geological Survey, Menlo Park, California.

Buer, K. 1984. Middle Sacramento River spawning gravel study. California Department of Water Resources, Northern District, Red Bluff.

Buer, K. 1994a. Sacramento River bank erosion investigation memorandum progress report. Internal memorandum to R. Scott and L. Brown from K. Buer, Chief, Geology Section, California Department of Water Resources, Northern District, Red Bluff.

Buer, K. 1994b. Use of alternative gravel sources for fishery restoration and riparian habitat enhancement in Shasta and Tehema counties, CA. Internal memorandum to J. Siperek, G. Stacey, and T. Mills from K. Buer, Chief, Geology Section, California Department of Water Resources, Northern District, Red Bluff. August 1.

Buer, K. 1995. Sacramento River gravel study: Keswick Dam to Cottonwood Creek. California Department of Water Resources, Northern District, Red Bluff.

Buer, K., D. Forwalter, M. Kissel, and B. Stohler. 1989. The middle Sacramento River: human impacts on physical and ecological processes along a meandering river. Pages 22–32 in D. L. Abell, editor. Proceedings of the California riparian systems conference: protection, management, and restoration for the 1990s. General Technical Report PSW-110. USDA Forest Service, Pacific Southwest Forest and Range Experiment Station, Berkeley, California.

CDWR (California Department of Water Resources). 1980. Upper Sacramento River spawning gravel study. Report. Prepared for California Department of Fish and Game by CDWR, Northern District, Red Bluff.

CDWR (California Department of Water Resources). 1981. Upper Sacramento River baseline study: hydrology, geology, and gravel resources. CDWR, Northern District, Red Bluff.

CDWR (California Department of Water Resources). 1999. Sacramento River aerial atlas: Colusa (River Mile 143) to Red Bluff (River Mile 243). Derived from WR-BNE-C 1:7,200 color photos flown on May 24, 1999. CDWR, Sacramento.

CDWR (California Department of Water Resources). 2002. Cow Creek to Jellys Ferry Bridge Geomorphic Baseline Investigation. Report prepared under direction of K. Buer and B. E. Ross.

Constantine, C., T. Dunne, and M. Singer. Unpublished. Controls on spatial differences in meander migration rates in a large gravel-bed river.

Dietrich, W. E., G. Day, and G. Parker. 1999. The Fly River, Papua New Guinea: inferences about river dynamics, floodplain sedimentation and fate of sediment. Pages 345–376 in A. J. Miller and A. Gupta, editors. Varieties of fluvial form. John Wiley & Sons, Chichester, England.

Dietrich, W. E., and J. D. Smith. 1983. Influence of the point bar on flow through curved channels. Water Resources Research 19: 1173–1192.

Dunne, T. 1998. Critical data requirements for prediction of erosion and sedimentation in mountain drainage basins. Journal of the American Water Resources Association 34: 795–808.

Furbish, D. 1988. River-bend curvature and migration: how are they related? Geology 16: 725–755.

Gay, G., H. Gay, W. Gay, and H. Martinson. 1998. Evolution of cutoffs across meander necks in Powder River, Montana, USA. Earth Surface Processes and Landforms 23: 651–662.

Greco, S. E., and R. E. Plant. 2003. Temporal mapping of riparian landscape change on the Sacramento River, miles 196–218, California, USA. Landscape Research 28: 405–426.

Greco, S. E., J. L. Tuil, and A. Wheaton. 2003. A historical aerial photography collection of the Sacramento River from Colusa to Red Bluff, California: 1937–1998. Prepared by Landscape Analysis and Systems Research Laboratory, Department of Environmental Design, University of California, Davis for the California Department of Water Resources, Northern District, Red Bluff.

Gurnell, A. M., S. R. Downward, and R. Jones. 1994. Channel planform change on the River Dee meanders, 1876–1992. Regulated Rivers: Research and Management 9: 187–204.

Ham, D. G., and M. Church. 2000. Bed-material transport estimated from channel morphodynamics: Chilliwack River, British Columbia. Earth Surface Processes and Landforms 25: 1123–1142.

Harvey, M. D. 1989. Meanderbelt dynamics of the Sacramento River, California. Pages 54–59 in D. L. Abell, editor. Proceedings of the California riparian systems conference: protection, management, and restoration for the 1990s. General Technical Report PSW-110. USDA Forest Service, Pacific Southwest Forest and Range Experiment Station, Berkeley, California.

Hasegawa, K. 1989. Universal bank erosion coefficient for meandering rivers. Journal of Hydraulic Engineering 115: 744–765.

Hickin, E. J., and G. C. Nanson. 1984. Lateral migration rates of river bends. Journal of Hydraulic Engineering 110: 1557–1567.

Hooke, J. M. 1980. Magnitude and distribution of rates of river bank erosion. Earth Surface Processes 5: 143–157.

Hooke, J. 1984. Changes in river meanders: a review of techniques and results of analyses. Progress in Physical Geography 8: 473–508.

Howard, A., and T. Knutson. 1984. Sufficient conditions for river meandering: a simulation approach. Water Resources Research 20: 1659–1667.

Ikeda, S., G. Parker, and K. Sawai. 1981. Bend theory of river meanders, Part 1: Linear development. Journal of Fluid Mechanics 112: 363–377.

Johannesson, H., and G. Parker. 1989. Linear theory of river meanders. Pages 181–213 in S. Ikeda and G. Parker, editors. River meandering. Water Resources Monograph 12. American Geophysical Union, Washington, D. C.

Klinesteker, S. 1998. Draft Sacramento River meander belt-future erosion study. Internal memorandum to K. Buer from S. Klinesteker, California Department of Water Resources, Northern District, Red Bluff. September 18.

Knighton, A. D. 1984. Fluvial forms and processes. Edward Arnold, London, United Kingdom.

Kondolf, G. M. 2000. Assessing salmonid spawning gravel quality. Transactions of the American Fisheries Society 129: 262–281.

Kondolf, G. M., T. Griggs, E. W.Larsen, S. McBain, M.Tompkins, J. G. Williams, and J. Vick. 2000. Flow regime requirements for habitat restoration along the Sacramento River between Colusa and Red Bluff. CALFED Bay Delta Program, Integrated Storage Investigation, Sacramento, California.

Larsen, E. W. 1995. Mechanics and modeling of river meander migration. Doctoral dissertation. University of California, Berkeley.

Larsen, E. W., A. K. Fremier, and S. E. Greco. Unpublished. Cumulative effective stream power and river channel migration on the Sacramento River, California, USA. *Submitted to Environmental Management.*

Larsen, E. W., E. H. Girvetz, and A. K. Fremier. 2006. Assessing the effects of alternative setback channel constraint scenarios employing a river meander migration model. Environmental Management 37: 880–897.

Larsen, E. W., and S. E. Greco. 2002. Modeling channel management impacts on river migration: a case study of Woodson Bridge State Recreation Area, Sacramento River, California, USA. Environmental Management 30: 209–224.

Larsen, E. W., E. H. Girvetz, and A. K. Fremier. 2006. Assessing the effects of alternative setback channel constraint scenarios employing a river meander migration model. Environmental Management 37: 880–897.

Lawler, D. M. 1993. The measurement of river bank erosion and lateral change: a review. Earth Surface Processes and Landforms 18: 777–821.

Leopold, L. B. 1994. A view of the river. Harvard University Press, Cambridge, Massachusetts.

Lewin, J. 1976. Initiation of bed forms and meanders in coarse-grained sediment. Geological Society of America Bulletin 87: 255–281.

Ligon, F. K., W. E. Dietrich, and W. J. Trush. 1995. Downstream ecological effects of dams: a geomorphic perspective. BioScience 45: 183–192.

MacDonald, T. E., G. Parker, and D. P. Leuthe. 1993. Inventory and analysis of stream meander problems in Minnesota. Saint Anthony Falls Hydraulic Laboratory, Minneapolis, Minnesota.

McCuddin, M. E. 1977. Survival of salmon and trout embryos and fry in gravel-sand mixtures. Master's thesis. University of Idaho, Moscow.

Micheli, E. R., and E. W. Larsen. In preparation. River channel cutoff dynamics, Sacramento River, California, USA.

Micheli, E. R., J. W. Kirchner, and E. W. Larsen. 2004. Quantifying the effect of riparian forest versus agricultural vegetation on river meander rates, central Sacramento River, California, USA. River Research and Applications 20: 537–548.

Morken, I., and G. M. Kondolf. 2003. Evolution assessment and conservation strategies for Sacramento River oxbow habitats. Prepared for The Nature Conservancy, Sacramento River Project. Berkeley, California.

Nanson, G., and E. Hickin. 1986. A statistical analysis of bank erosion and channel migration in western Canada. Bulletin Geological Society of America 97: 497–504.

Piégay, H., G. Bornette, A. Citterio, E. Hérouin, B. Moulin, and C. Statiotis. Channel instability as a control on silken dynamics and vegetation patterns within perifluvial aquatic zones. Hydrologic Processes 14: 3011–3029.

Pizzuto, J. E. and T. S. Meckelnburg. 1989. Evaluation of a linear bank erosion equation. Water Resources Research 25: 1005–1013.

RCE (Resource Consultants & Engineers, Inc.). 1992. Riverbed gradient restoration Sacramento River mile 206 California: Design memorandum. Prepared by Resource Consultants & Engineers, Inc. for USACE (US Army Corps of Engineers).

Reiser, D. W., and R. G. White. 1988. Effects of two sediment size-classes on survival of steelhead and chinook salmon eggs. North American Journal of Fisheries Management 8: 432–437.

Schumm, S., and M. Harvey. 1986. Preliminary geomorphic evaluation of the Sacramento River: Red Bluff to Butte Basin. Prepared for Army Corps of Engineers by Water Engineering and Technology, Fort Collins, Colorado.

Sommer, T. R., M. L. Nobriga, W. C. Harrell, W. Batham, and W. J. Kimmerer. 2001a. Floodplain rearing of juvenile Chinook salmon: evidence of enhanced growth and survival. Canadian Journal of Fisheries and Aquatic Sciences 58: 325-333.

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

Stillwater Sciences. 2005. Sacramento River ecological flow study–gravel study plan. Prepared for the Nature Conservancy by Stillwater Sciences.

Thomas, J. 2000. A physical model of meander migration on the central Sacramento River. Master's thesis. University of California, Davis.

Thorne, C. R. 1992. Bend scour and bank erosion on the meandering Red River, Louisiana. Pages 95–115 in P. Carling and G. Petts, editors. Lowland floodplain rivers: geomorphological perspectives. Wiley and Sons, Chichester.

USACE (US Army Corps of Engineers). 1981. Sacramento River and tributaries bank protection and erosion control investigation California—Study of alternatives.

USACE. 1991. Sloughs and tributaries. Sacramento River aerial atlas: Collinsville to Shasta Dam. USACE, Sacramento, California.

US Congress. 1960. Sacramento flood control project, California: Report from the Chief Engineers, US Army, US 86th Congress, 2nd Session. Senate Document 103: 19–29

WET (Water Engineering and Technology). 1988. Geomorphic analysis of the Sacramento River. Draft report DACWO5-87-C-0084. Prepared by Water Engineering and Technology, Fort Collins, Colorado for Army Corps of Engineers.

WET (Water Engineering and Technology). 1990. Geomorphic analysis of the Sacramento River. Phase II Report. Prepared by Water Engineering and Technology, Fort Collins, Colorado for Army Corps of Engineers.

Winterbottom, S. J., and D. J. Gilvear. 2000. A hydrological and geomorphological based approach to mapping and predicting river bank erosion within a GIS framework: River Tummel, Scotland. Regulated Rivers: Research and Management 16: 127–140.

Figures

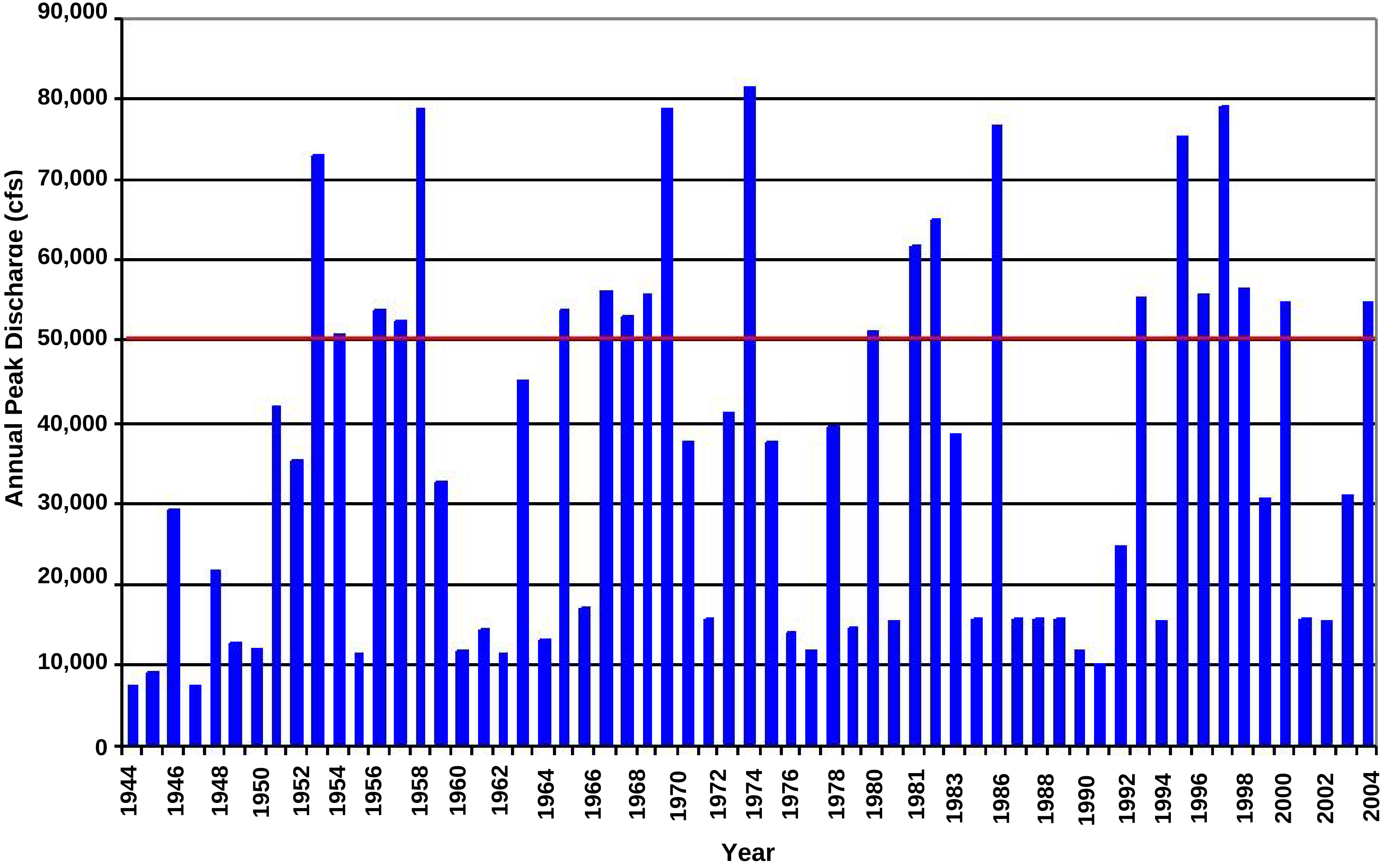


Figure 3-1. Annual peak discharge versus time for the period 1944-2004. The estimated threshold for bed scouring flow events below Keswick Dam is 50,000 cfs (red line). Since the completion of Shasta Dam in 1945, the USGS gauge at Keswick (no. 11370500) has registered several floods with magnitudes greater than 50,000 cfs.

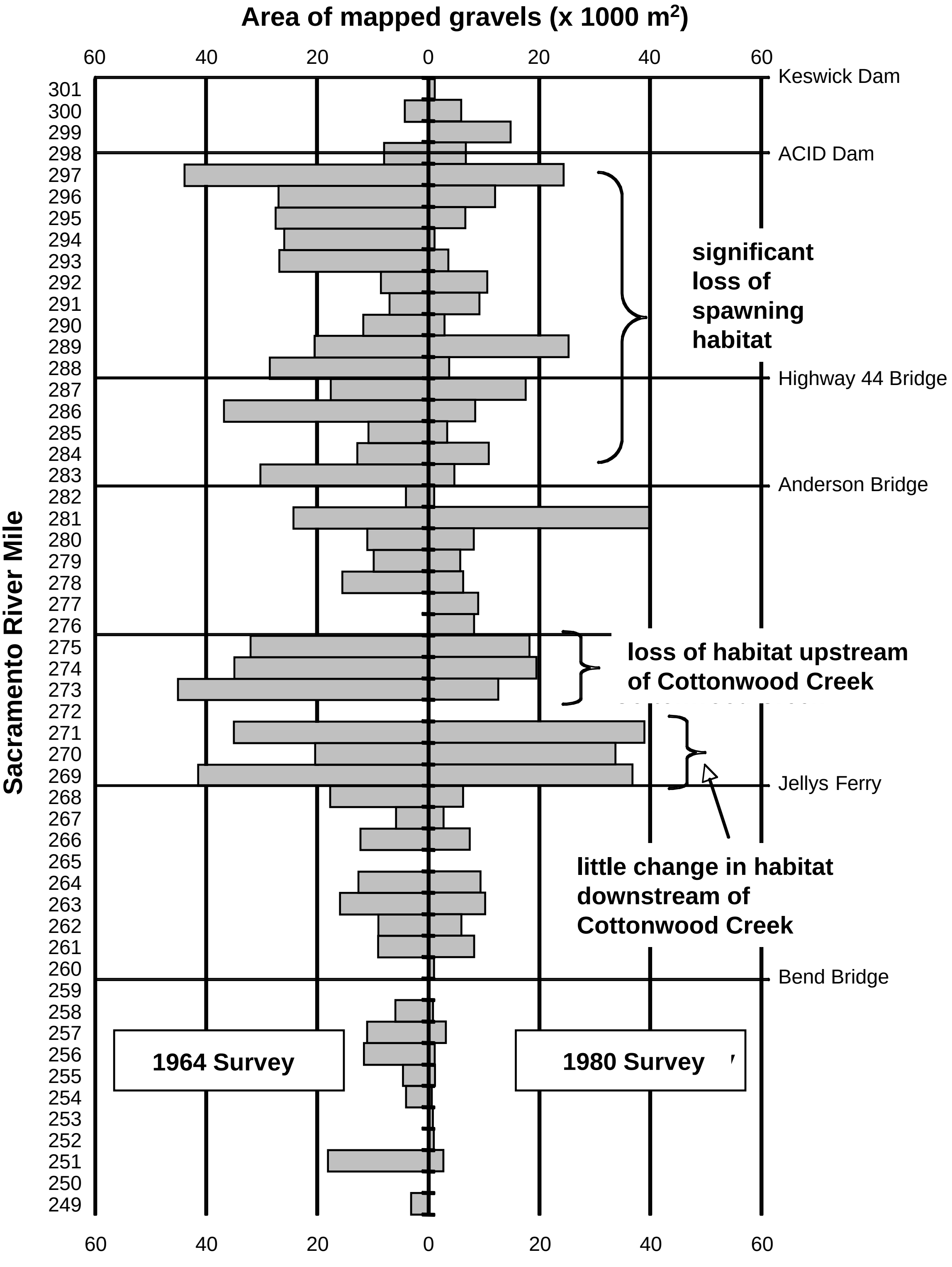


Figure 3-2. Change in spawning habitat over time. Bars on the left of the zero axis represent spawning habitat area by river mile as mapped by the 1964 survey; bars on the right of the axis show the spawning habitat area by river mile derived from the 1980 survey. Bed coarsening may have reduced spawning habitat between RM 298.4 and RM 283. The persistence of spawning habitat downstream of Cottonwood Creek illustrates the effects of locally high sediment supply.

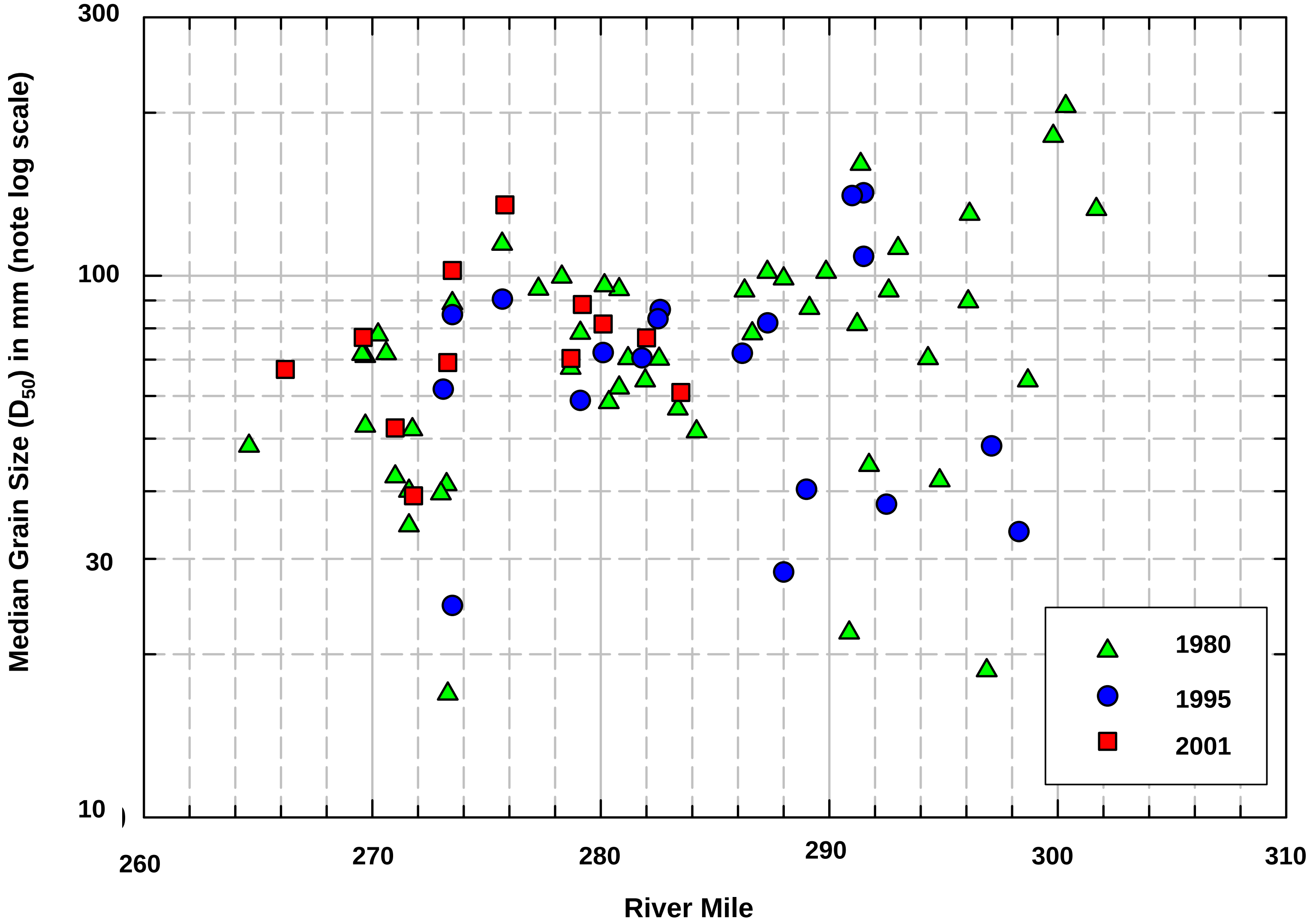


Figure 3-3. Time series of median grain sizes (determined from Wolman counts) as a function of river mile for geomorphically comparable positions on point bars. Overlap among data from successive sample efforts, together with wide scatter within each data set, are consistent with essentially no change in median grain size over time, and substantial natural variability in grain size at the local (i.e., few tens to hundreds of meters) scale.

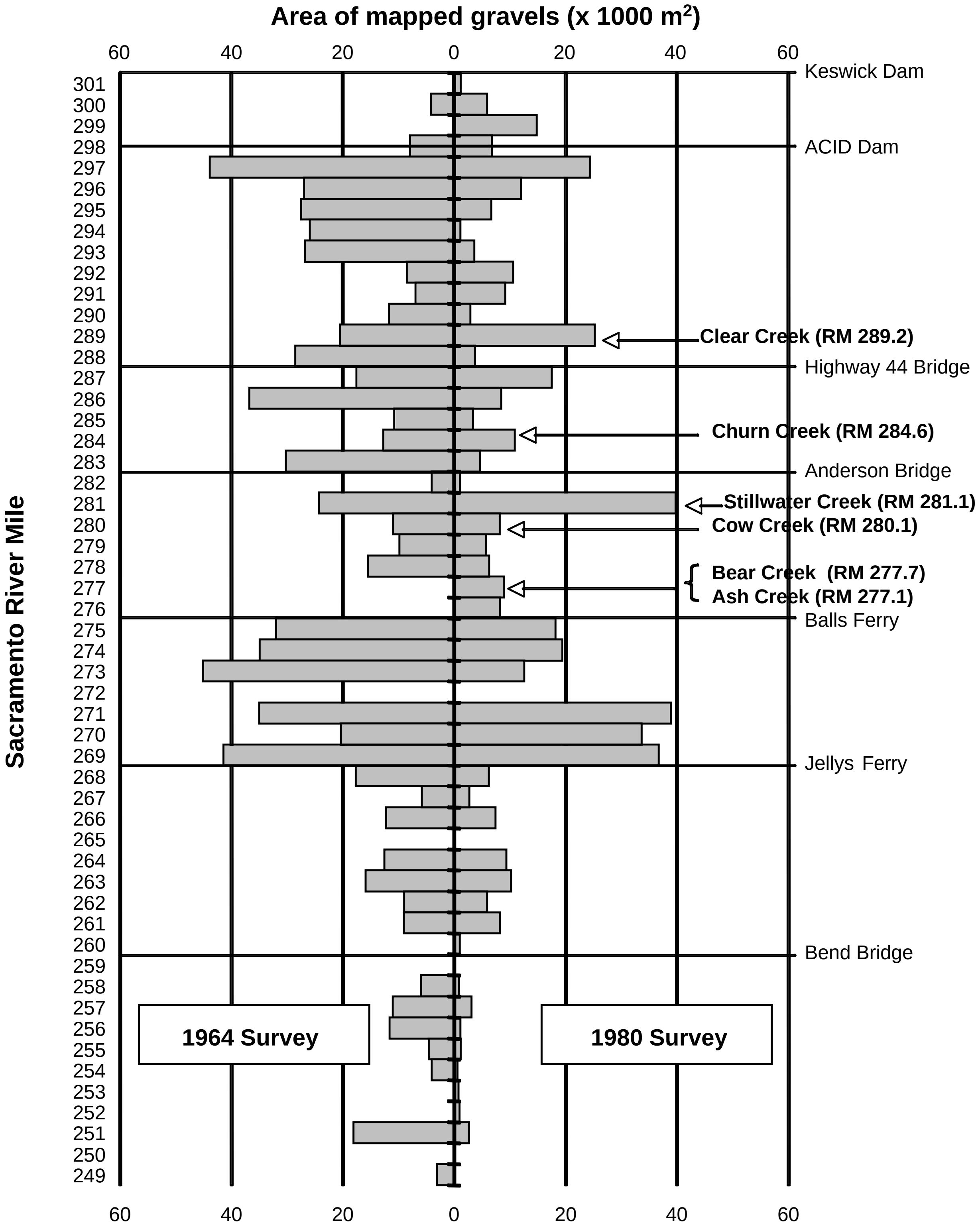


Figure 3-4. Change in spawning habitat, with emphasis on local effects of gravel additions. Bars on the left of the zero axis represent spawning habitat area by river mile as mapped by the 1964 survey; bars on the right of the axis show the spawning habitat area by river mile derived from the 1980 survey. Tributaries bear small amounts of sediment, helping to maintain patches of spawning habitat near their confluences with the mainstem. This produces localized effects, with spawning habitat downstream largely unaffected. Gravel augmentation is likewise localized and may create a similar pattern of effects on spawning gravel.

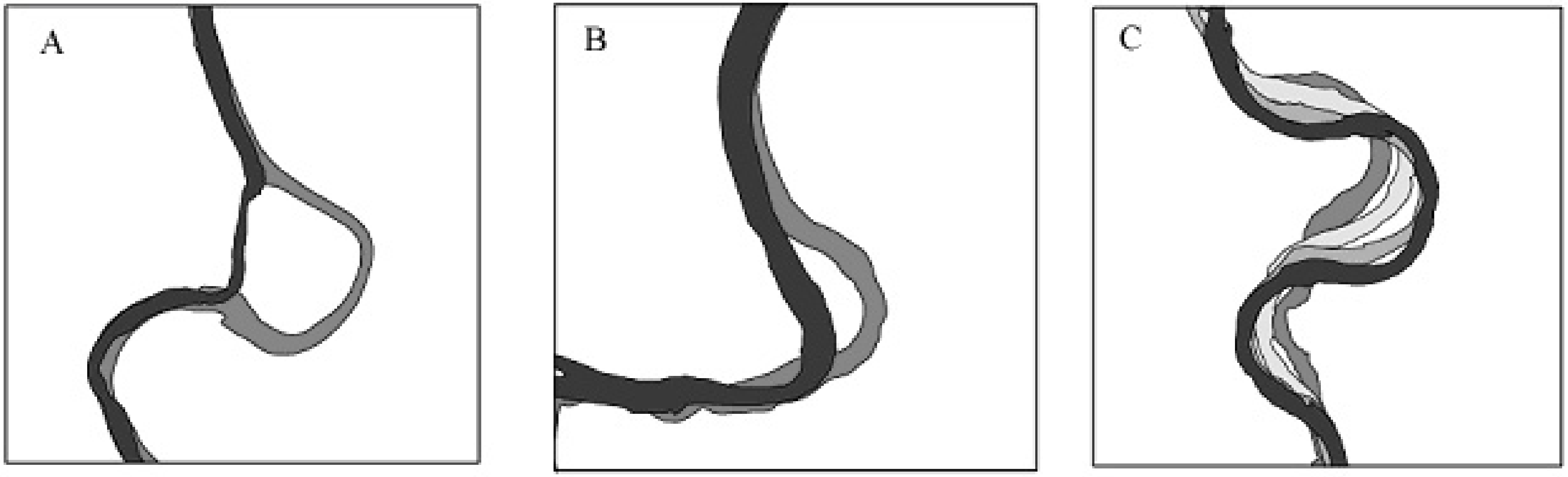


Figure 3-5. Styles of channel migration. A. Chute cutoff: excavation of a secondary channel (darkly shaded), which crosses the floodplain and provides a shortcut for capture of nearly all of the river's flow (also called a "complete" cutoff). B. Partial cutoff: channel avulsion (darkly shaded) that affects only a portion of a bend and may create a persistent mid-channel bar and a slow flowing side channel (also called a "neck" cutoff). C. Progressive migration: differences in shading show downstream migration of a meander bend due to a gradual process of lateral bank erosion and point bar deposition.

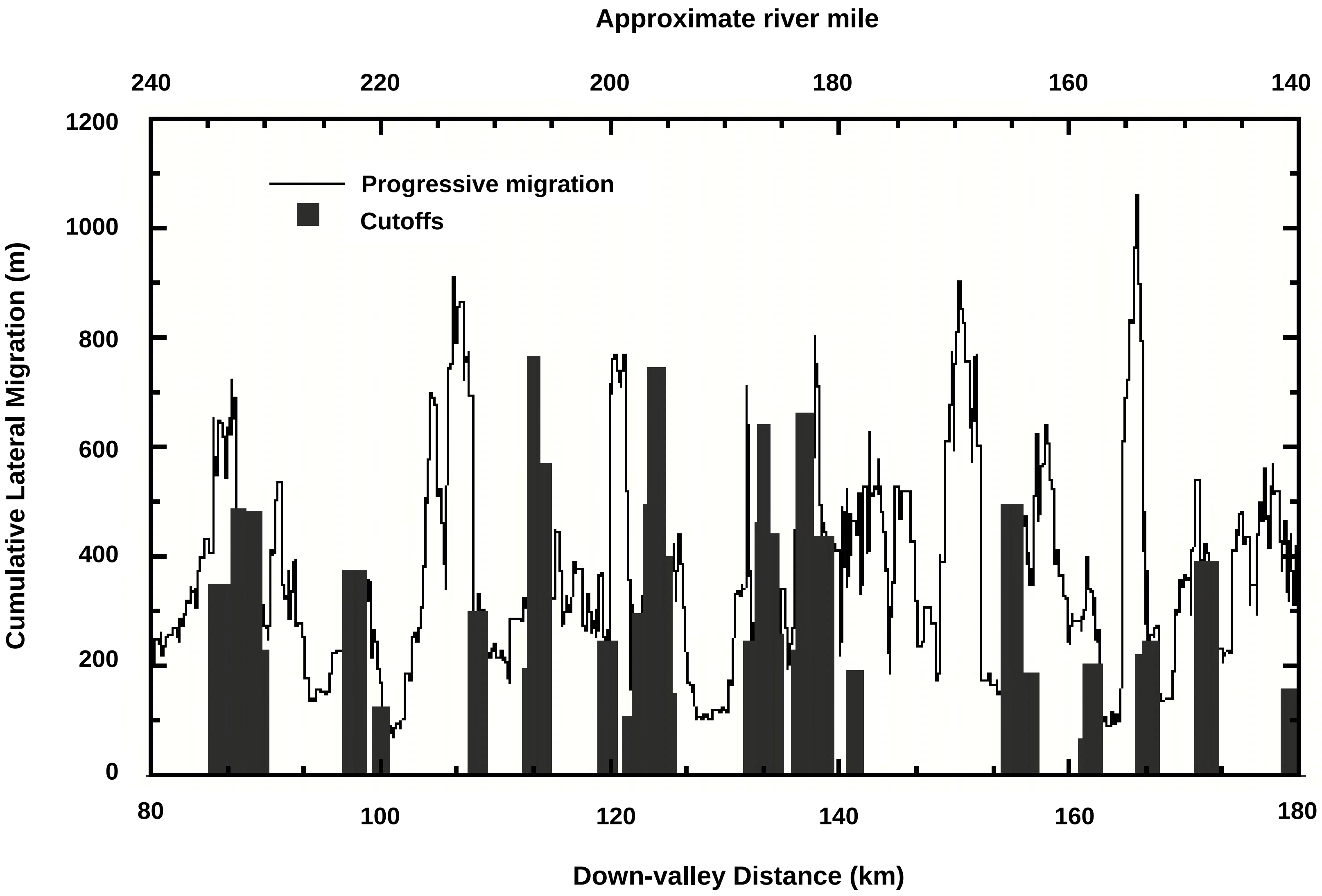


Figure 3-6. Cumulative meander migration from 1904–1997 on the central Sacramento River, for progressive migration (thin line) and cutoffs (vertical bars). Troughs and peaks show areas of stability and instability respectively. Source: Micheli and Larsen. (in preparation).

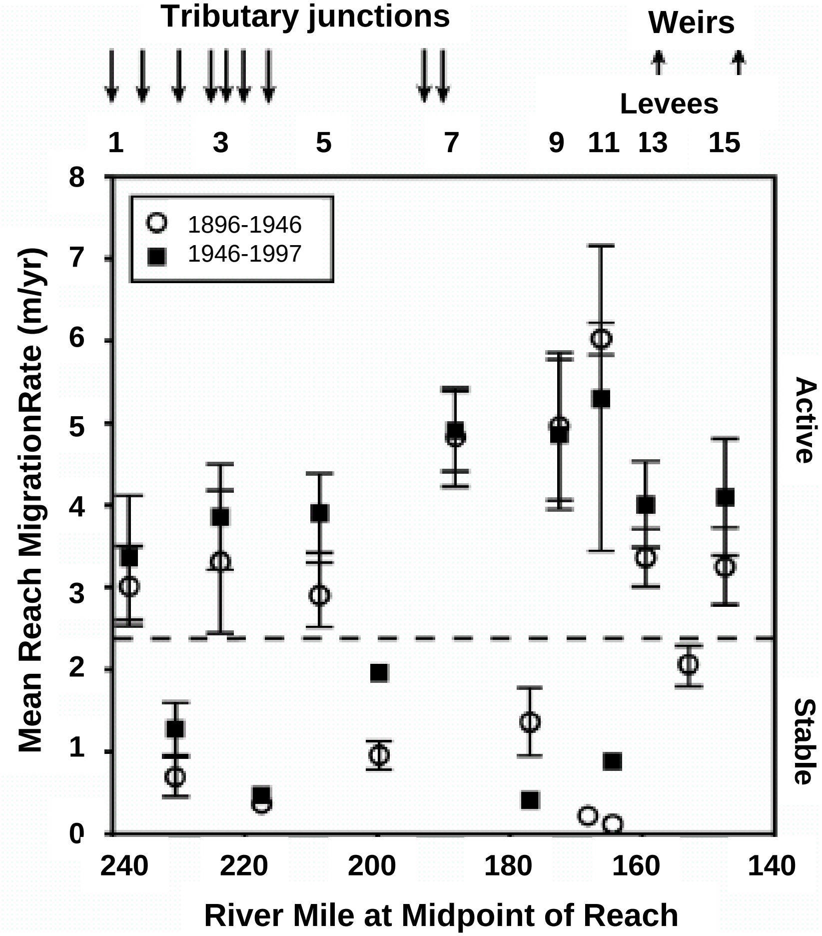


Figure 3-7. Rates of channel migration of the central Sacramento River averaged by reach (with reach designations—from Schumm and Harvey 1986—listed on the upper axis), showing "active" and "stable" reaches classified relative to threshold migration rate of 2.2 m/yr (7.2 ft/yr). Post-dam migration rates (closed symbols) generally plot higher than pre-dam rates (open symbols), but error bars usually overlap, implying that differences are not statistically significant. Locations of major weirs, levees, and tributary junctions are shown on the upper axis. Source: Constantine et al. *in review***.**

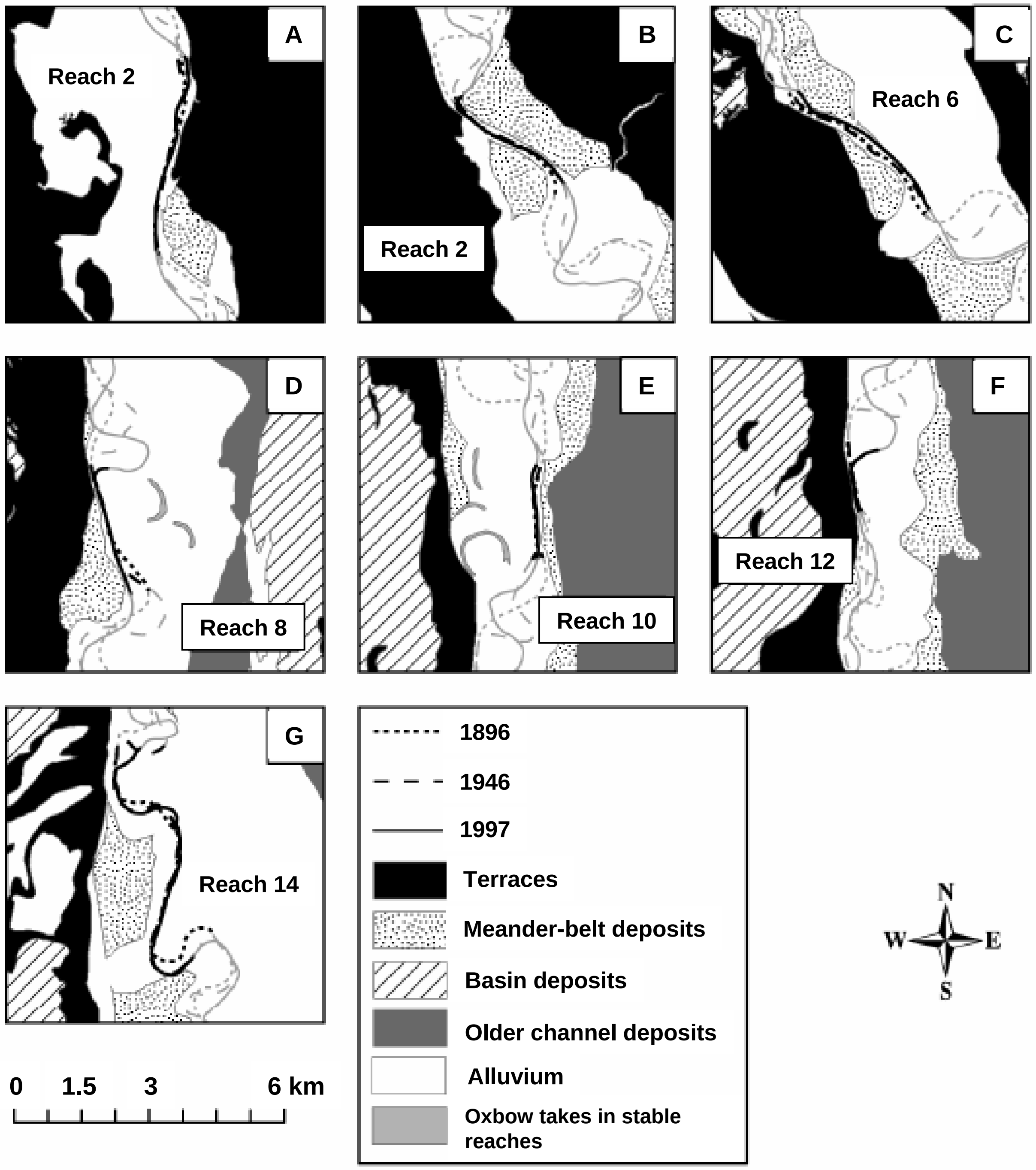


Figure 3-8. Maps of surface geology and the evolution of stable reaches from 1896 to 1997. Thick black lines delineate stable reaches. "Terraces" include Tehama, Red Bluff, Riverbank, and Modesto Formations and small areas underlain by ash beds. "Meander-belt deposits" are undifferentiated colluvium aged 100 to 1000 years. "Basin deposits" are fine-grained material of unknown age. "Older channel deposits" are coarse Pleistocene paleochannel gravels. "Alluvium" includes material in the 100-year meander belt, point bar scrolls, tributary channel deposits, and other undifferentiated alluvium. Source: Constantine et al. ***in review*****.**

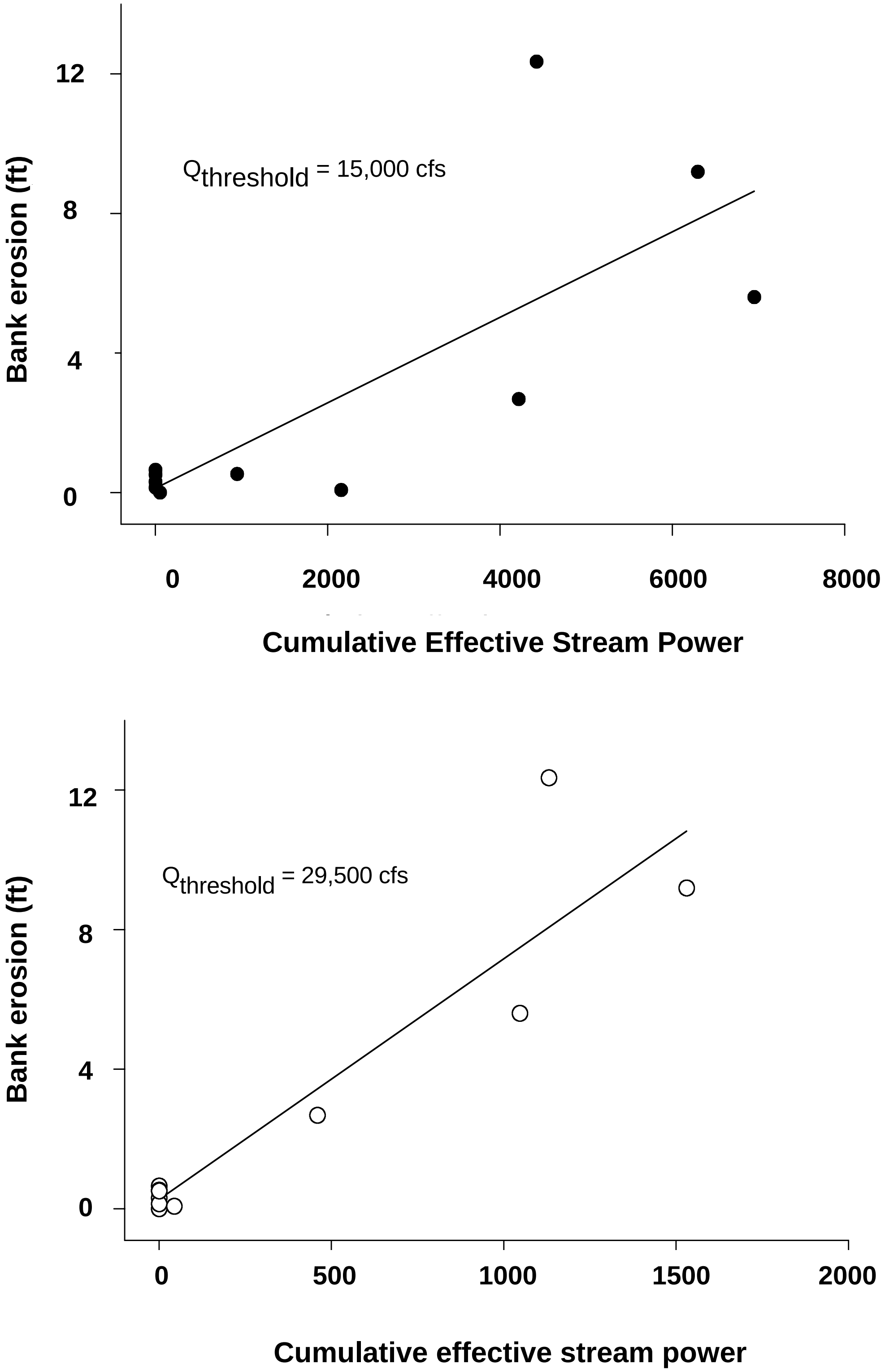


Figure 3-9. Annual bank erosion plotted against cumulative effective stream power for two threshold discharges: 15,000 cfs in upper plot and 29,000 cfs in lower plot. (Reprinted from Kondolf et al. 2000).

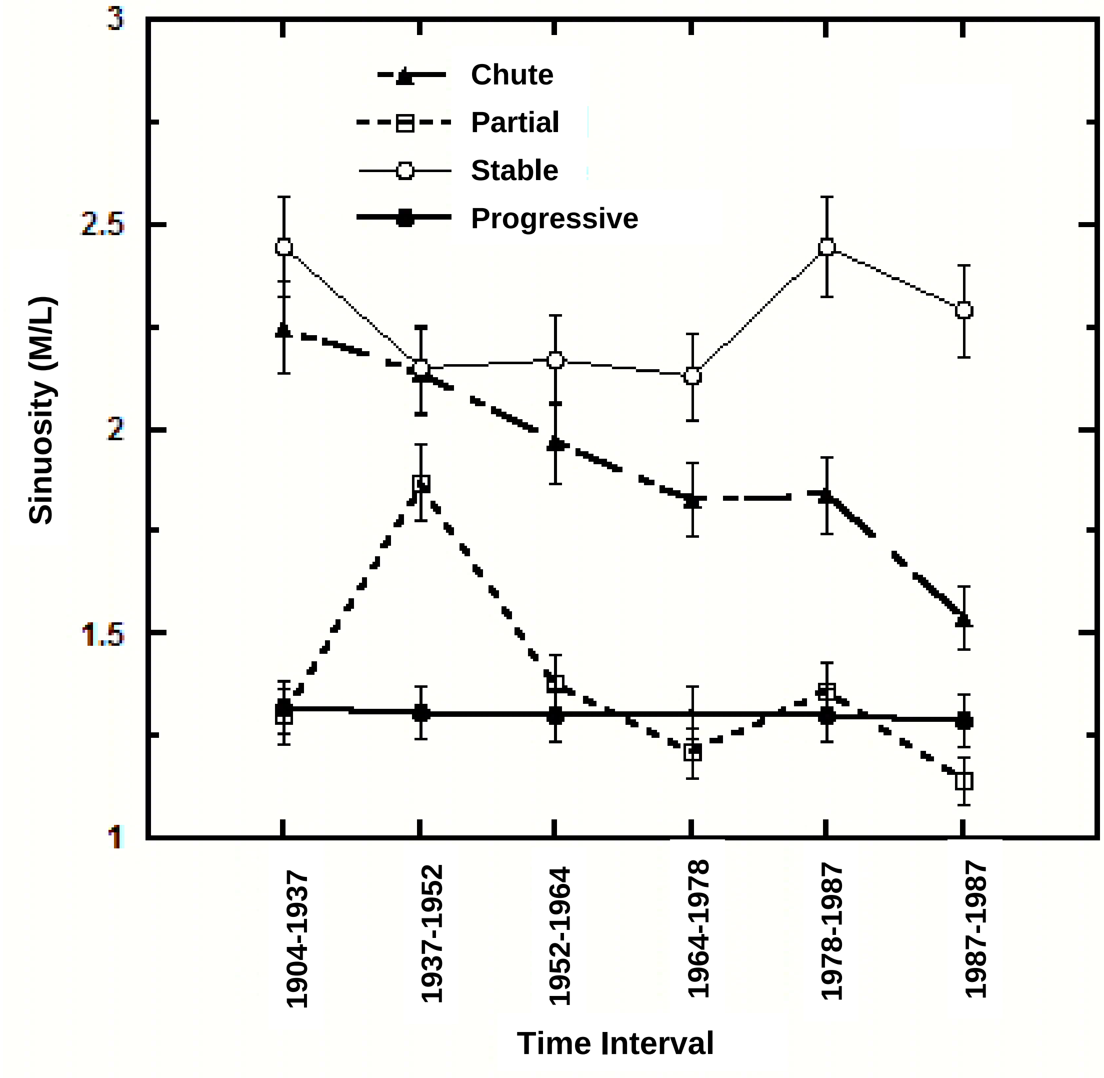


Figure 3-10. Average sinuosity of chute cutoffs (triangles), partial cutoffs (open squares), progressive migration (closed squares), and stable bends (circles) on the central Sacramento River for six time intervals between 1904–1997. Bends affected by chute cutoffs show a systematic decrease in average sinuosity over time. The average sinuousity of progressively migrating bends has been roughly stable over the period of record. Source Micheli et al. ***in review*.**

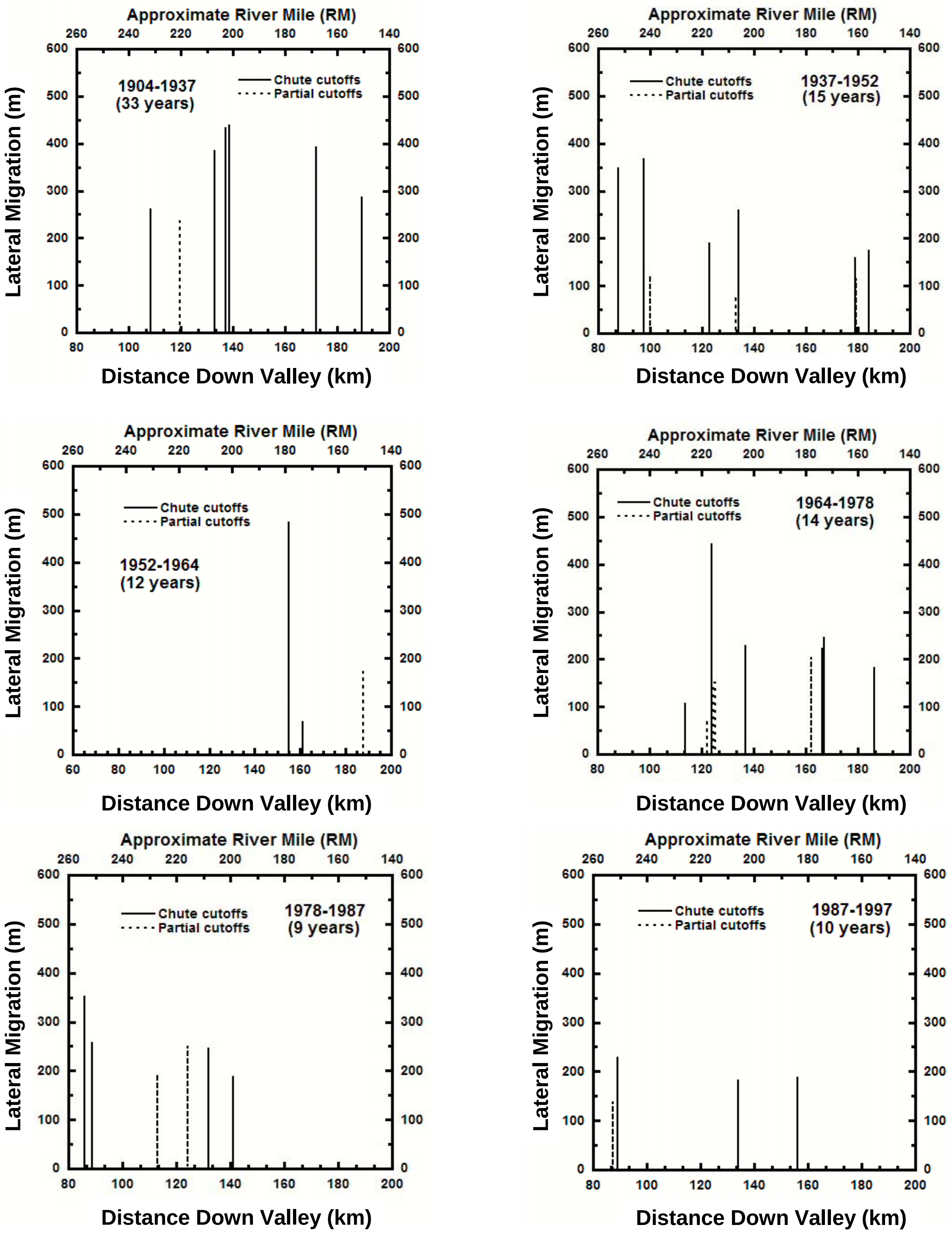


Figure 3-11. Location, magnitude, and timing of cutoffs, middle Sacramento River, for six time steps from 1904-1997. A general pattern of decreasing cutoff size is apparent, particularly in the last two intervals. This is consistent with data presented in Table 3-10. Source: Micheli et al. ***in review*****.**

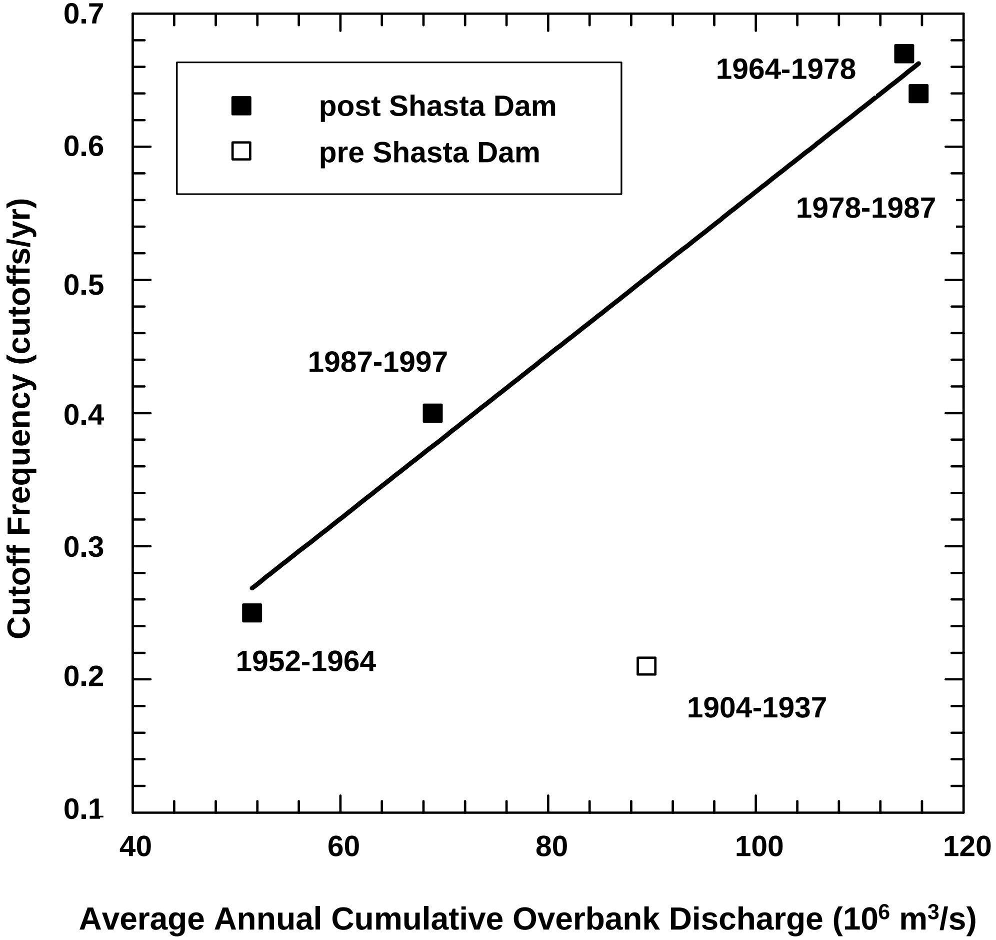


Figure 3-12. Cutoff frequency versus average annual cumulative overbank discharge on the middle Sacramento River for 5 intervals. A roughly linear relationship (blank line) between cutoff frequency and overbank flow is apparent for the four intervals that post-date construction of Shasta Dam (closed symbols). The fact that the sole data point for the pre-dam interval (open symbol) is an outlier suggests that the pre-dam era may have been marked by a fundamentally different relationship. Such a difference could be due to effects of bank stabilization projects (i.e., riprap installation, which was progressive throughout the mid to late 20th-century) or a fundamental difference in erodibility of the floodplain. On the other hand the discrepancy may reflect limitations of the data; the possibility that the number of cutoffs was underestimated for the pre-dam interval is difficult to rule out, given that the older basemaps probably had lower resolution than those that were used to estimate cutoff frequencies for later intervals.
Source: Micheli et al. *in review*.

4 CHINOOK SALMON

4.1 Chinook salmon habitat requirements

The Sacramento River basin is the largest watershed in California (~27,000 mi^2) and empties into the largest estuary on the west coast of the United States. This diverse basin is unique in that it supports four runs of Chinook salmon, including the winter-run, which only occurs in the Sacramento basin. Because the four runs exhibit a variety of different life-history strategies, anthropogenic activities in the basin have affected each of the runs differently. The habitat requirements and the life-history strategies of the four runs are discussed below and differential impacts of anthropogenic effects are discussed in the following sections of this chapter.

4.1.1 Upstream migration and holding

Adult Chinook salmon require water deeper than 0.8 ft (24 cm) and water velocities less than 8 ft/s (2.4 m/s) for successful upstream migration (Thompson 1972). Adult Chinook salmon appear to be less capable of negotiating fish ladders, culverts, and waterfalls during upstream migration than coho salmon or steelhead (Nicholas and Hankin 1989), due in part to slower swimming speeds and inferior jumping ability compared to steelhead (Reiser and Peacock 1985; Bell 1986). The maximum jumping height for Chinook salmon has been calculated to be approximately 7.9 ft (2.4 m) (Bjornn and Reiser 1991).

Both winter-run and spring-run Chinook salmon return to the Sacramento River when reproductively immature, typically holding for a few months in deep pools near spawning areas until spawning. Adult Chinook salmon require large, deep pools with flowing water for summer holding. Adult Chinook tend to hold in pools with depths > 4.9 ft (> 1.5 m) that contain cover from undercut banks, overhanging vegetation, boulders, or woody debris (Lindsay et al. 1986), and have water velocities ranging from 0.5 to 1.2 ft/s (15 to 37 cm/s) (Marcotte 1984). Water temperatures for adult Chinook holding are reportedly best when < 60.8°F (< 16°C), and lethal when > 80.6°F (> 27°C) (Moyle et al. 1995). Spring Chinook in the Sacramento River system typically hold in pools below 69.8–77°F (21–25°C).

4.1.2 Spawning

Most Chinook salmon spawn in larger rivers or tributaries, although spawning has been observed in streams as small as 7–10 ft (2–3 m) in width (Vronskiy 1972). Chinook typically spawn in low- to moderate-gradient reaches of streams, but can navigate shorter reaches with steeper gradients to access suitable spawning areas. Armantrout (in press, as cited by ULEP 1998) concluded that Chinook salmon seldom inhabit streams with gradients > 3% after examining extensive inventory data from Oregon. The upper extent of Chinook distribution in the Umpqua River basin, Oregon appears to occur where gradients are less than 3% (ULEP 1998).

Upon arrival at the spawning grounds, adult females dig shallow depressions or pits in suitably sized gravels (discussed in further detail below), deposit eggs in the bottom during the act of spawning, and cover them with additional gravel. Over a period of one to several days, the female gradually enlarges the redd by digging additional pits in an upstream direction (Burner 1951). Redd areas vary considerably depending on female size, substrate size, and water velocities, and can range from 5.4 ft^2 (0.5 m^2) (Nelson and Banford 1983, as cited in Healey 1991) to 482 ft^2 (44.8 m^2) (Chapman et al. 1986, as cited in Healey 1991).

Chinook salmon tend to seek spawning sites with high rates of intergravel flow. Upwelling, which is associated with a concave bed profile, may be an important feature selected for by spawning Chinook salmon (Vaux 1968).

Chinook are capable of spawning within a wide range of water depths and velocities, provided that intergravel flow is adequate for delivering sufficient oxygen to eggs and alevins (Healey 1991). Depths most often recorded over Chinook redds range from 4 to 80 in (10 to 200 cm) (Burner 1951, Chambers et al. 1955, Vronskiy 1972) and velocities from 0.5 to 3.3 ft/s (15 to 100 cm/s) (Burner 1951, Chambers et al. 1955, Thompson 1972, Vronskiy 1972, Smith 1973), although values may vary between races and stream basins. Fall Chinook salmon, for instance, are able to spawn in deeper water with higher velocities such as the mainstem Sacramento River, because of their larger size (Hallock et al. 1957, as cited in Healey 1991).

Substrate particle size composition has been shown to have a significant influence on intragravel flow dynamics (Platts 1979). Chinook salmon may therefore have evolved to select redd sites with specific particle size criteria that will ensure adequate delivery of dissolved oxygen to their incubating eggs and developing alevins. In addition, salmon are limited by the size of substrate that they can physically move during the redd building process. Substrates selected likely reflect a balance between water depth and velocity, substrate composition and angularity, and fish size. As depth, velocity, and fish size increase, Chinook are able to displace larger substrate particles. D_{50} values (the median diameter of substrate particles found within a redd) for spring Chinook have been found to range from 10.8 mm to 78.0 mm (0.43 in to 3.12 in) (Platts et al. 1979, Chambers et al. 1954, 1955, all as cited in Kondolf and Wolman 1993).

In 1997, USFWS researchers collected data on substrate particle size, velocity, and depth at hundreds of Chinook salmon redds in the Sacramento River between Keswick Dam and Battle Creek to develop habitat suitability criteria for use in models that can aid in determining instream flows beneficial for anadromous salmonids. Redds in both shallow and deep areas were sampled. The following table summarizes habitat suitability criteria data collected in this study for three of the four runs (too few spring-run redds were found from which to collect data). Much more detail on the methods used and results can be found in USFWS (2003).

Table 4.1-1. Range of suitable habitat values for Chinook salmon spawning in the Sacramento River (USFWS 2003).

Run	Range of Suitable Values					
	Velocity		Depth		Substrate	
	ft/s	m/s	ft	m	in	cm
Fall	0.93–2.66	0.28–0.81	1–14	0.3–4	1–3 to 3–5	3–8 to 8–13
Late-fall	0.90–2.82	0.27–0.86	1–14	0.3–4	1–3 to 4–5	3–8 to 10–13
Winter	1.54–4.10	0.47–1.25	3–16	0.9–5	1–3 to 3–5	3–8 to 8–13

4.1.3 Egg incubation and alevin development

Once redd construction is completed, a key determinant of survival from egg incubation through fry emergence is the amount of fine sediment in the gravel (McCuddin 1977, Reiser and White

1988). High concentrations of fine sediment in (or on) a streambed in a redd can reduce permeability and intergravel flow. This can result in reduced delivery rate of oxygen and increasingly elevated metabolic waste levels around incubating eggs, larvae, and sac-fry as they develop within egg pockets (Kondolf 2000), which can in turn lead to high mortality. Several studies have correlated reduced dissolved oxygen levels with mortality, impaired or abnormal development, delayed hatching and emergence, and reduced fry size at emergence in anadromous salmonids (Wickett 1954, Alderdice et al. 1958, Coble 1961, Silver et al. 1963, McNeil 1964a, Cooper 1965, Shumway et al. 1964, Koski 1981). Silver et al. (1963) found that low dissolved oxygen concentrations are related to mortality and reduced size in Chinook salmon and steelhead embryos. Fine sediments in the gravel interstices can also physically impede fry emergence, trapping (or entombing) them within the redd (Phillips et al. 1975, Hausle and Coble 1976).

The effects of high fine sediment concentrations may be counteracted to a certain extent by the redd construction process itself. As adult salmon build their redds, they displace fine material downstream and coarsen the substrate locally (Kondolf et al. 1993, Peterson and Foote 2000, Moore et al. 2004). However, the effects of sediment reduction during redd construction may be rapidly reversed by infiltration of fine sediment into the redds during the incubation period (Kondolf et al. 1993).

Suitable water temperatures are required for proper embryo development and emergence. Incubating eggs of Chinook can withstand constant temperatures between 35.1°F (1.7°C) (Combs and Burrows 1957) and 62.1°F (16.7°C) (USFWS 1999); however, substantial mortality may occur at the extremes. Myrick and Cech (2004) conclude that temperatures between 43 and 54°F (6°C and 12°C) are best for ensuring egg and alevin survival. Sublethal stress and/or mortality of incubating eggs resulting from elevated temperatures would be expected to begin at temperatures of about 58°F (14.4°C) for constant exposures (Combs and Burrows 1957, Combs 1965, Healey 1979).

The most recent study conducted on Sacramento River Chinook salmon egg temperature tolerance was conducted by the USFWS (1999, as cited in Myrick and Cech 2004). This study showed fall-run Chinook salmon egg mortality increasing at temperatures greater than 53.6°F (12°C) and winter-run egg mortality increasing at temperatures greater than 56.0°F (13.3°C) (Myrick and Cech 2004). Higher mortality in the post-hatching period was observed at higher temperatures, as was also found by Healey (1979, as cited in Myrick and Cech 2001). According to Myrick and Cech (2001), the study suggested “that winter-run eggs and fry may be slightly more tolerant of elevated temperatures than fall-run”; however, the results generally agree with those found for populations in more northern regions, and there does not appear to be much variation, if any, with regard to egg thermal tolerances between runs of Chinook salmon (Healey 1979, Myrick and Cech 2001).

4.1.4 Rearing

4.1.4.1 Fry rearing

Following emergence, fry occupy low-velocity, shallow areas near stream margins, including backwater eddies and areas associated with bank cover such as large woody debris (Lister and Genoe 1970, Everest and Chapman 1972, McCain 1992). As the fry grow, they tend to move into deeper and faster water further from banks (Hillman et al. 1987, Everest and Chapman 1972, Lister and Genoe 1970). The work of Everest and Chapman (1972) suggests that habitat with water velocities less than 0.5 ft/s (15 cm/s) and depths < 24 in (60 cm) are suitable for newly emerged fry.

Although fry typically drift downstream following emergence (Healey 1991), movement upstream or into cooler tributaries following emergence has also been observed in some systems (Lindsay et al. 1986, Taylor and Larkin 1986). On the Sacramento River, juvenile Chinook salmon are more commonly found in association with natural (as opposed to riprapped) riverbanks, and Shaded Riverine Aquatic (SRA) cover (CDFG 1983; Michny and Hampton 1984; Michny and Deibel 1986; Michny 1987, 1988, 1989; Fris and DeHaven 1993). DeHaven (1989, as cited in Fris and DeHaven 1993) found that the distribution of juvenile Chinook was less tied to riparian habitat features when low water temperatures prevailed, but that association with SRA cover increased with increased temperatures (over 70°F [21°C]).

4.1.4.2 Juvenile rearing

Little is known regarding habitat selection of juvenile Chinook salmon in the Sacramento River system. Habitat preferences of Chinook salmon may vary depending on the type of channel available (e.g., constrained vs. unconstrained), substrate and bank characteristics, abundance of small and large wood, presence of other salmonids (particularly coho salmon), and whether the Chinook are of ocean- or stream-type. In addition, habitat use has been observed to change between seasons, between night and day, and over time as fish grow and can occupy habitats with higher water velocities.

Several researchers have found relationships between velocity and juvenile Chinook habitat use, with juveniles generally occupying areas with water velocities less than 15–30 cm/s (Thompson 1972, Hillman et al. 1987, Steward and Bjornn 1987, Murphy et al. 1989, Beechie et al. 2005), that have cover such as brush, large wood, undercut banks (Hillman et al. 1987, Johnson et al. 1992, Demko et al. 1998, Beechie et al. 2005). Lister and Genoe (1970) found that juvenile Chinook salmon preferred “slow water adjacent to faster water (40 cm/s),” and Shirvell (1994) suggested that preferred habitat locations vary by activity. For feeding, they are likely to select positions with optimal velocity conditions, whereas for predator avoidance, optimal light conditions are more likely to be important (Shirvell 1994). At night, juvenile Chinook appear to move to quiet water or pools and settle to the bottom, returning the next day to the riffle and glide habitats they had occupied the previous day (Edmundson et al. 1968, Don Chapman Consultants 1989).

Although some researchers have found juvenile Chinook to reside primarily in pools, they may also use glides and runs, as well as riffles. Where coho salmon are present, the two species appear to exhibit stronger niche separation, with juvenile coho salmon using low-velocity pools, and juvenile Chinook salmon using swifter habitats such as glides, runs, and riffles, or deeper water farther from shore and cover (Taylor 1991). In the absence of coho salmon, Chinook may prefer deeper pools with low water velocities during spring and summer as well as winter (Lister and Genoe 1970, Everest and Chapman 1972, Swales et al. 1986, Hillman et al. 1987). In the Elk River, Oregon, Burnett and Reeves (2001) found most juvenile ocean-type Chinook salmon (in sympatry with coho salmon and steelhead) in valley segments with deeper pools, larger volume pools, and pools with greater densities of large wood. In Elk River tributaries, the juveniles were observed almost exclusively in pools. Roper et al. (1994) also found age-0+ Chinook to be strongly associated with pools in the South Umpqua River basin, Oregon. In the Sacramento and American rivers, CDFG (1997) found juvenile Chinook salmon densities to be highest in runs, closely followed by pools, with fish also occupying riffles and glides.

Temperatures have a significant effect on juvenile Chinook growth rates. On maximum daily rations, growth rate increases with temperature to a certain point and then declines with further increases. Reduced rations can also result in reduced growth rates; therefore, declines in juvenile salmonid growth rates are a function of both temperature and food availability. Laboratory studies indicate that juvenile Chinook salmon growth rates are highest at rearing temperatures from 65–70°F (18.3–21.1°C) in the presence of unlimited food (Clarke and Shelbourn 1985, Banks et al. 1971, Brett et al. 1982, Rich 1987), but decrease at higher temperatures. Myrick and Cech (2004) note that two studies have been published on the relationship between temperature and growth of Central Valley Chinook salmon—one by Marine and Cech (2004) on Sacramento River fall-run Chinook, and one by Myrick and Cech (2002) on American River fall-run Chinook. Provided food is not limiting, these studies showed that optimum temperatures for growth were between 63 and 68°F (17 and 20°C). Under natural conditions, it is unlikely that Chinook salmon will feed at 100% rations, and disease, competition, and predation are also factors that may affect survival. In order to determine temperatures that might be optimal for growth of juvenile Chinook under natural conditions, Brett et al. (1982) used a value of 60% rations, based on field studies that suggested fish in the wild fed at roughly 60% of their physiological maximum. When used in a model developed for sockeye salmon, he determined that juvenile Chinook salmon would reach their optimal growth at a temperature of about 59°F (15°C) (Brett et al. 1982, as cited in Myrick and Cech 2004). Nicholas and Hankin (1989) suggest that the duration of freshwater rearing is tied to water temperatures, with juveniles remaining longer in rivers with cool water temperatures.

Temperatures of > 74.0°F (23.3°C) are considered potentially lethal to juvenile Chinook salmon (Hanson 1990). Myrick and Cech (2004) recently summarized available information on juvenile Chinook salmon temperature tolerances. Incipient upper lethal temperature (IULT) studies, which may be the most biologically relevant for studying juvenile temperature tolerances, are lacking for Central Valley Chinook salmon. Sacramento River fall-run Chinook were reared at temperatures between 70 and 75°F (21 and 24°C) by Marine and Cech (2004) without significant mortality; however, Rich (1987) observed significant mortality after only 8 days of rearing at 75°F (24°C) (Myrick and Cech 2004). Myrick and Cech (2004) suggest that, until IULT studies are conducted on Central Valley Chinook, managers use Brett's (1952) and Brett et al.'s (1982) data on more northern Chinook, which determined that the IULT is in the range of 24–25°C (75–77°F). More detail on temperature tolerances of various Chinook life stages can be found in Myrick and Cech (2001, 2004).

4.1.4.3 Winter rearing

Juvenile Chinook salmon rearing in tributaries may disperse downstream into mainstem reaches in the fall and take up residence in deep pools with LWD, in interstitial habitat provided by boulder and rubble substrates, or along river margins (Swales et al. 1986, Healey 1991, Levings and Lauzier 1991). During high flow events, juveniles have been observed to move to deeper areas in pools and they may also move laterally in search of slow water (Shirvell 1994, Steward and Bjornn 1987). Hillman et al. (1987) found that individuals remaining in tributaries to overwinter chose areas with cover and low water velocities, such as areas along well-vegetated, undercut banks. There is very little information available on Chinook salmon use of floodplains and off-channel habitats such as sloughs and oxbows compared to coho salmon. However, recent studies in the Sacramento and Cosumnes rivers have shown that shallow seasonally inundated floodplains can provide suitable rearing habitat for Chinook (Sommer et al. 2001a; P. Moyle, pers. comm., as cited in Sommer et al. 2001a).

In winter, juvenile Chinook salmon may make use of the interstitial spaces between coarse substrates as cover (Bjornn 1971, Hillman et al. 1987). Hillman et al. (1987) found that the addition of cobble substrate to heavily-sedimented glides in the fall substantially increased winter rearing densities, with Chinook using the interstitial spaces between the cobbles as cover. Fine sediment can act to reduce the value of gravel and cobble substrate as winter cover by filling interstitial spaces between substrate particles. This may cause juvenile Chinook to avoid these embedded areas and move elsewhere in search of suitable winter cover (Stuehrenberg 1975, Hillman et al. 1987).

Over much of the Chinook salmon's range, winter temperatures are too cold to allow for much growth in the winter. The low-temperature threshold for positive growth in juvenile Chinook salmon is believed to be about 40.1°F (4.5°C), with 39.4°F (4.1°C) being the lower limit for zero net growth in a juvenile Chinook population (Armour 1990, as cited in McCullough 1999). In the Sacramento River, water temperatures rarely fall below 43°F (6°C); however, allowing for growth throughout the winter.

Few researchers have focused on the habitat criteria of juvenile Chinook salmon during the winter. However, both Hillman et al. (1987) and Shirvell (1994) conducted quality research that directly pertains to selection of habitat criteria. Based on their studies, habitat with maximum water velocities of 0.66 ft/s (20 cm/s) can be considered suitable for juvenile Chinook in winter.

In the Sacramento/San Joaquin system some juvenile Chinook salmon rear on seasonally inundated floodplains in the winter. Sommer et al. (2001a) found higher growth and survival rates of Chinook juveniles that reared on the Yolo Bypass floodplain than in the mainstem Sacramento River, and Moyle (2000) observed similar results on the Cosumnes River floodplain. On the Yolo Bypass, bioenergetic modeling suggested that increased prey availability on the floodplain was sufficient to offset increased metabolic demands from higher water temperatures (9°F [5°C] higher than mainstem). Sommer et al. (2001a) believe that the well-drained topography (e.g., floodplains with few pits and depressions) may help reduce stranding risks when flood waters recede.

4.1.5 Smoltification and outmigration

In many systems, juvenile Chinook spend up to several months in estuaries, feeding and growing before entering the ocean (Healey 1991). Juveniles of all four runs of Chinook salmon in the Central Valley must pass through the Sacramento-San Joaquin Estuary (Delta) on their way to the ocean and many rear there for varying periods prior to ocean entry. Substantial numbers of fry may be found in the Delta from January through March; relatively few have been found in other months in the 20 years of sampling from 1977 to 1997 (Brandes and McLain 2001). The annual abundance of fry (defined as < 2.8 in [70 mm] fork length [FL]) during this period appears related to flow, with the highest numbers observed in wet years (Brandes and McLain 2001).

Although growth rates of juvenile Chinook salmon may be high at temperatures approaching 66°F (19°C), cooler temperatures may be required for Chinook to successfully complete the physiological transformation from parr to smolt. Smoltification in juvenile Sacramento River fall-run Chinook was studied by Marine (1997, as cited in Myrick and Cech 2001), who found that juveniles reared under a high temperature regime of 70–75°F (21–24°C) exhibited altered and impaired smoltification patterns relative to those reared at low 55–61°F (13–16°C) and moderate 63–68°F (17–20°C) temperatures. Some alteration and impairment of smoltification was also seen in the juveniles reared at the moderate temperatures. Chronic exposure to high temperatures may also result in greater vulnerability to predation. In this same study by Marine (1997), Sacramento River fall-run Chinook salmon reared at the highest temperatures 70–75°F (21–24°C) were preyed upon by striped bass more often than those reared at low or moderate temperatures. Consumption rates of piscivorous fish such as Sacramento pikeminnow, striped bass, and largemouth bass increase with temperature, which may compound the effects of high temperature on juvenile and smolt predation mortality. Juvenile growth rates are an important influence on survival because juvenile salmon are gape-limited predators that are themselves subject to gape-limited predation by larger fish. Faster growth thus both increases the range of food items available to them and decreases their vulnerability to predation (Myrick and Cech 2004).

4.2 Winter-Run Chinook Salmon

Though Chinook salmon range from California's Central Valley to Alaska and the Kamchatka Peninsula in Asia, winter-run Chinook are only found in the Sacramento River. Chinook of this race are unique because they spawn during the summer months when air temperatures usually approach their yearly maximum. As a consequence, winter-run Chinook require stream reaches with cold water sources that will protect embryos and juveniles from the warm ambient conditions in the summer. Historically, high-elevation reaches of tributaries to the upper Sacramento River (e.g., McCloud River) provided the cold water reaches that supported summer spawning by winter-run Chinook. Currently, hypolimnetic releases from Shasta Lake provide the cold water temperatures that allow winter-run Chinook to persist downstream of the dam, despite the complete loss of historical spawning habitat, access to which was cut off upon completion of the dam. Winter-run Chinook are also unique because the construction and operation of Shasta Dam probably contributed to an initial **increase** in the population soon after dam construction, primarily by creating more spawning habitat with suitable water temperatures than was available prior to dam construction. However, the positive effect of Shasta Dam on winter-run Chinook began to wane in the late 1970s and early 1980s, when escapements reached dangerously low levels. The population crash stimulated consideration of winter-run Chinook as a threatened species beginning in 1985 when the California-Nevada chapter of the American Fisheries Society petitioned NMFS to list the run as a threatened species (AFS 1985). Following a dangerously low year-class in 1989, NMFS issued an emergency listing for Sacramento River winter-run Chinook salmon as a threatened species in 1989 (NMFS 1989); the California Fish and Game Commission listed winter-run as endangered in the same year. After several years of low escapements in the early 1990s, the status of winter-run was changed from threatened to endangered by NMFS in 1994 (NMFS 1994).

As the lead sub-chapter in the Chinook section, this discussion of winter-run Chinook salmon generally includes more detail than the successive sub-chapters addressing spring-run, fall-run, and late-fall-run Chinook salmon. Many of the factors that influence the winter-run salmon population in the Sacramento River also similarly affect the other runs, so many of the analyses discussed in this sub-chapter are relevant to the remaining Chinook runs.

4.2.1 Distribution

Winter-run Chinook salmon are found only in the Sacramento River basin. The distribution of winter-run Chinook spawning has shifted over time in response to changes in upstream passage caused by water supply development and operations. Prior to construction of Shasta Dam in the 1940s, winter-run Chinook salmon spawned in the upper Sacramento River system (Little

Sacramento, McCloud, and possibly in the Pit and Fall rivers) and in nearby Battle Creek (Yoshiyama et al. 1998). Since the construction of Shasta Dam, winter-run Chinook have been limited to the mainstem Sacramento River below Keswick Dam (RM 302), though a few adults occasionally stray into tributaries (e.g., Battle and Mill creeks) to spawn (Harvey-Arrison 2001). The distribution of spawning likely shifted again in 1966, when the construction and operation of Red Bluff Diversion Dam (RBDD) (RM 243.5) impeded access to upstream reaches, forcing more winter-run adults to spawn downstream of the diversion dam. A radio-tag survey of winter-run adults between 1979 and 1981 indicated that adults were delayed at RBDD between 1 and 40 days, with an average delay of 18 days (Hallock and Fisher 1985). The dam also forced winter-run adults to spawn downstream of Red Bluff, where summer water temperatures were frequently too high to support successful egg incubation and emergence. Beginning in 1986, the U.S. Bureau of Reclamation (USBR) began raising RBDD gates during the winter to facilitate upstream passage of winter-run Chinook (Table 4.2-1) (USBR 2004), which precipitated an upstream shift in the distribution of winter-run spawning.

Until 2001, most winter-run spawning occurred downstream of ACID Dam (RM 298.4), but an improvement of this dam’s fish passage facilities in 2001 allowed another upstream shift in the distribution of spawning (Figure 4.2-1) (CDFG 2002a, 2004).

Table 4.2-1. Periods when RBDD gates were raised to facilitate upstream passage of winter-run Chinook salmon and the percentage of spawning located downstream of RBDD (1987–2003) (Source: USBR 2004).

Year	Winter-run % spawning below RBDD	Months RBDD gates raised
1987	5	December–March
1988	25	December–mid-February
1989	2	December–mid-April; gates in 11 days in February
1990	7	December–March
1991	0	December–April
1992	4	December–April
1993	2	15 September–15 May
1994	0	15 September–15 May
1995	1	15 September–15 May
1996	0	15 September–15 May
1997	0	15 September–15 May
1998	3	15 September–15 May
1999	0	15 September–15 May
2000	0	15 September–15 May
2001	0.4	15 September–15 May
2002	0.2	15 September–15 May
2003	0.3	15 September–15 May

4.2.2 Population trends

There is little historical data available to characterize winter-run Chinook escapements prior to the construction of Shasta Dam; indeed, the agencies did not recognize winter-run Chinook salmon as a distinct run until the 1940s (Needham et al. 1943). In the late 1930s, the pending construction of Shasta Dam prompted the agencies to commission a study of potential salmon salvage options. As part of this investigation, researchers placed a counting weir at ACID Dam between 1937 and 1939 to estimate the size of the salmon run in the Sacramento River (Hatton et al. 1940). The counting weir enabled scientists to estimate the run size of the fall-run salmon populations; however, the removal of flashboards from the ACID Dam during winter prevented observations of winter-run salmon during their period of upstream migration (December-May).

The first reference to winter-run salmon in the Sacramento River was made by personnel from the Baird Hatchery, which was located on the McCloud River near the confluence with the Sacramento River before the site was inundated by Shasta Reservoir. They observed a single adult salmon spawning in the McCloud River on 24 April 1902, which was too early to be a spring-run salmon (Rutter 1904). A more substantial observation of winter-run salmon occurred 37 years later. Researchers participating in the Shasta Dam salmon salvage investigation observed salmon spawning "on the upper McCloud River between Lower Falls and Big Springs during May and June 1939," including 25 adult salmon on 26 May 1939 (Hatton et al. 1940). The timing of the observed spawning correlates with the life history timing of winter-run Chinook salmon. This observation, coupled with the reports of sportsmen and ranchers of "a winter or 'black' run of Salmon in the Sacramento River" led Hatton et al. (1940) to entertain "the possible existence of a third run of salmon" in 1940. Though the evidence was building for the existence of a winter-run of Chinook salmon in the Sacramento River, there was no attempt to enumerate the population.

Subsequent observations of winter-run salmon in the Sacramento River were derived from the incidental capture and transport of adults as part of the spring-run salmon salvage operation in the mid-1940s. Shasta Dam construction began to block upstream passage in May 1942, so researchers began capturing spring-run salmon at Keswick in 1943 for transfer to Deer Creek as part of the salmon salvage plan. During the trap-and-haul operation, researchers observed ripe females in the tanker trunks in June 1943, indicating winter-run salmon were mixed with the un-ripe spring-run adults that had been captured (Needham et al. 1943). Later in June 1943, they also observed spawned-out carcasses in Deer Creek where transferred salmon had been planted, further indicating the presence of winter-run salmon. Needham et al. (1943) estimated that 59 adult winter-run had been captured at Keswick and transferred to Deer Creek, but only seven of these winter-run salmon survived to spawn in Deer Creek. The winter-run salmon spawning was likely unsuccessful in Deer Creek, because water temperatures in July and August 1943 were probably lethal to incubating eggs (Slater 1963). There were no direct observations or estimates of winter-run salmon spawning in the mainstem Sacramento River in 1943. However, any spawning that did occur was likely unsuccessful because water temperatures below the Shasta Dam construction site were probably too warm for successful incubation of eggs, given that they were considered lethal for spring-run adults (Moffett 1949). Consequently, the winter-run population likely suffered a complete year-class failure in 1943.

In 1944, no observations of winter-run salmon were noted in the transfer of salmon from the Sacramento River to Deer Creek as part of the spring-run salvage operations, nor were there any direct observations of winter-run spawning in the mainstem river (Slater 1963). However, Slater (1963) hypothesized that summer water temperatures in the mainstem Sacramento River near Redding (estimated between 52° and 61°F [11.1° and 16.1°C] in June and July) may have permitted some successful spawning of winter-run salmon in 1944. On the other hand, Slater (1963) also noted that poor water quality, caused by metal contamination of Shasta Reservoir releases, killed adult fish in the Sacramento River in November 1944. Winter-run adults may have suffered similar mortality from poor water quality during their holding and spawning periods earlier in 1944. The combination of adult mortality from metal contamination and egg mortality from summer water temperatures likely limited production from the 1944 year class of winter-run salmon, and it may have caused another year-class failure.

In 1945, nearly 200 winter-run adults were observed in the transfer of salmon from Keswick to Deer Creek but, again, elevated summer water temperatures in Deer Creek were likely lethal to any incubating eggs of winter-run salmon that spawned in the creek (Slater 1963). In contrast, the estimated water temperatures in the mainstem Sacramento River near Redding ranged between 46°F (7.8°C) and 50°F (10°C) in June and July of 1945, which led Slater (1963) to hypothesize that winter-run spawned successfully in the mainstem Sacramento River that year.

There were no direct observations of winter-run salmon spawning in the mainstem Sacramento River between 1943 and 1946—the first years when the construction of Shasta Dam blocked upstream passage. Nevertheless, incidental observations of winter-run salmon during trap-and-haul operations for spring-run salmon, coupled with poor environmental conditions in both the Sacramento River and Deer Creek, led Slater to conclude that "the winter-run populations were small" in the years when Shasta Dam was being constructed (1963).

Slater (1963) hypothesized that the winter-run salmon population began to rebound in 1947, and that "this initial recovery seems to have been both substantial and rapid" from the "low point of 1943–1946." He cites an angling survey conducted by Smith (1950), which evaluated the 1947–1948 and 1949–1950 sport fishery in the upper Sacramento River. "Increased catches of winter-run Chinook salmon in January and February 1949" (Slater 1963) led Smith (1950, as cited in Slater 1963) to conclude that a "sizable" winter-run population existed. Similarly, Slater cited an increase in the number of winter-run salmon that were harvested by Coleman National Fish Hatchery between 1949 and 1956 (as part of the fall-run salmon propagation program) (Azevedo and Parkhurst 1958, as cited in Slater 1963) as evidence that winter-run salmon escapements increased in the late 1940s and early 1950s. Though these qualitative assessments do not permit a detailed tracking of winter-run salmon abundance, they do suggest a positive trend in the population in the years after Shasta Dam was completed.

This positive trend seems to have continued through the 1950s, because Hallock estimated that 11,000 winter-run adults were harvested from the Sacramento River by anglers in the winter of the 1961–1962 fishing season (R. J. Hallock, pers. comm., as cited in Slater 1963). Hallock's estimate of the percentage of winter-run Chinook caught in the in-river recreational harvest suggests that total winter-run escapements in the winter of 1961–1962 numbered in the tens of thousands. In June 1963, Slater personally observed winter-run Chinook salmon spawning in the vicinity of Redding in numbers that approached the fall-run population that spawned in the same sites (Slater 1963). For context, the four years before Slater's observation of winter-run spawning in 1963 (1959–1962) had fall-run salmon escapement estimates ranging between 115,500 and 250,000 salmon. Though Slater observed spawning in only a small portion of the habitat available to both winter-run and fall-run salmon in the Sacramento River, his observation suggests that the winter-run salmon population had increased substantially from the few hundred fish captured during the trap-and-haul salvage operation in 1943 and 1945. His observation also suggests that the winter-run salmon population had recovered from a probable year-class failure in 1943 and a partial year-class failure in 1944.

Beginning in 1967, agency biologists began estimating annual winter-run escapements by monitoring adults migrating through the fish passage facilities of Red Bluff Diversion Dam.

Though the dam facilitated a more accurate account of the winter-run population, gate operations interfered with upstream passage. Gate operations were modified beginning in the winter of 1986 to facilitate the upstream passage of winter-run Chinook. However, raising the dam gates rendered winter-run escapement estimates less reliable, because migrating salmon could bypass the dam's fish counting facilities. Figure 4.2-2 displays the annual escapement estimates of winter-run Chinook between 1967 and 2005, which are derived primarily from RBDD counts.

The RBDD counts permitted agency biologists to track the decline in winter-run Chinook abundance beginning in the 1970s. The drought of 1976–1977 caused a precipitous decline in abundance between 1978 and 1979, when escapements fell below 2,500 fish. Though the population rebounded briefly to more than 20,000 fish in 1981, escapements ranged from the low hundreds to a few thousand fish throughout the 1980s and the mid-1990s. The population has been increasing since 2001, which prompted NMFS to propose re-classifying Sacramento River winter-run Chinook salmon from an endangered species to a threatened species in 2004 (NMFS 2004a).

4.2.3 Life history

Table 4.2-2 illustrates life history timing for winter-run Chinook salmon in the Sacramento River basin. Winter-run Chinook salmon display a life history that is intermediate between ocean-type and stream-type Chinook. They spend between five and ten months rearing in fresh water before migrating to sea, which is longer than typical ocean-type Chinook and shorter than typical stream-type Chinook salmon (Healey 1991).

Table 4.2-2. Life history timing of winter-run Chinook in the Sacramento River basin.

Life stage	Month																							
	Jan		Feb		Mar		Apr		May		Jun		Jul		Aug		Sept		Oct		Nov		Dec	
Adult entry into San Francisco Bay[1]	▒	▒	▒	▒	▒	▒	▒	▒	▒	▒	▒	▒									▒	▒	▒	▒
Migration past Red Bluff Diversion Dam[2]	▒	▒	▒	▒	▓	▓	▒	▒	▒	▒	░	░	░	░									░	░
Spawning[3]								▒	▓	▓	▓	▓	▒	▒	▒									
Incubation[3]								▒	▒	▒	▒	▒	▒	▒	▒	▒	▒	▒	▒					
Fry emergence[3]												▒	▒	▒	▒	▒	▒	▒	▒					
Rearing in lower reaches (age 0+)	▒	▒	▒	▒	▒	▒	▒	▒						▒	▒	▒	▒	▒	▒	▒	▒	▒	▒	▒
Outmigration past Red Bluff Diversion Dam[3]	▒	▒	▒	▒	▒								░	░	░	░	▓	▓	▓	▓	▒	▒	▒	▒
Entry into Sacramento-San Joaquin Delta	▒	▒	▒	▒	▒	▒	▒	▒																

[1] Van Woert (1958), Hallock et al. (1957), both as cited in NMFS (1997)
[2] Hallock and Fisher (1985)
[3] Vogel and Marine (1991)

░	Period of Light Activity
▒	Period of Moderate Activity
▓	Period of Peak Activity

4.2.3.1 Adult upstream migration and spawning

Adult winter-run Chinook enter San Francisco Bay from November through June (Van Woert 1958, Hallock et al. 1957, as cited in NMFS 1997). Migration past Red Bluff Diversion Dam (RBDD) begins in mid-December and can continue into early August, but the majority of winter-run adults migrate past RBDD between January and May, with a peak in mid-March (Hallock and Fisher 1985). Current RBDD operations facilitate upstream passage of winter-run adults by raising gates between 15 September and 15 May, which encompasses the vast majority of the upstream migration period for winter-run salmon. Since the change in RBDD gate operations, volitional spawning below RBDD by winter-run salmon is negligible in most years. The winter run appears to move upstream much more quickly than the spring run (Moyle et al. 1989, as cited in NMFS 1997), which may reflect an adaptation to historical flow conditions in the Sacramento River. Winter-run migrate upstream during a period when high flows are typically driven by rainfall events. As a result, the faster migration rate probably allowed winter-run to ascend high elevation reaches during relatively flashy flow events in the winter. In comparison, the spring-run salmon migrated upstream during the more prolonged snowmelt period in the late spring, so adults faced no penalty for migrating more slowly.

Like spring-run Chinook salmon, winter-run Chinook enter spawning streams while still reproductively immature. Adults hold for a few months in deep pools near spawning areas, which provides time for gonad development. Winter-run salmon spawn in late spring/early summer. This life-history strategy reduces competition for spawning habitat with other runs. However, it also makes the run reliant on year-round coldwater sources, which limits the potential for expanding the range of the run in the Sacramento River basin.

4.2.3.2 Juvenile rearing and outmigration

Winter-run fry emerge from the spawning gravels from mid-June through mid-October (NMFS 1997). Because winter-run salmon spawning is concentrated upstream in the reaches below Keswick Dam, the entire Sacramento River can serve as a nursery area for juvenile winter-run Chinook as they migrate downstream. Downstream movement of juveniles typically begins in August soon after fry emerge from redds. Rotary screw traps at RBDD usually record peaks in the abundance of winter-run salmon fry in September and October. However, following these initial pulses of fry, winter-run juveniles steadily stream past RBDD through March (Kimmerer and Brown, in prep.). Most juvenile winter-run Chinook reach the Delta between January and April, when they can pose a conflict with Delta pumping operations designed to increase South-of-Delta storage during winter months when conflicts with protections for Delta smelt are reduced.

4.2.4 Habitat requirements

General habitat requirements for Chinook salmon are described in Section 4.1. The winter-run Chinook salmon's life history is unique to the Sacramento River because it provides the thermal conditions that allow for the success of this strategy. Because winter-run Chinook spawn in late spring and early summer, they require access to stream reaches with summer water temperatures cool enough to allow egg incubation. The spawning reaches and reaches downstream must also have sufficiently warm water temperatures in the winter to support growth throughout this season because juveniles must grow large enough to smolt and outmigrate before water temperatures become too high the following spring and summer.

4.2.5 Conceptual model of historical population dynamics

This conceptual model highlights the implications of the late-spring/early-summer spawning strategy used by winter-run Chinook salmon and is intended to illustrate potential limiting factors (Figure 4.2-3).

4.2.5.1 Adaptation to the Sacramento River water temperature regime

Because winter-run Chinook spawn in late spring and early summer, their progeny emerge in late summer and early fall. No other salmonids in the Sacramento River emerge during this time, and most other juvenile salmonids outmigrate in the spring before summer water temperatures in the middle and lower Sacramento River become too warm. As a result, winter-run Chinook fry and juveniles have relatively little competition for rearing habitat in the fall and winter as they migrate downstream.

Salmon fry can tolerate warmer water temperatures than eggs or alevins; therefore, the amount of rearing habitat available to them includes not only the spawning areas, but also farther downstream wherever water temperatures are still suitable for rearing in the late summer and early fall. As water temperatures cool in the fall, progressively more of the mainstem river system becomes suitable as juvenile rearing habitat. In the Sacramento River, juvenile winter-run salmon are currently found in the reach above Deer Creek (RM 220) in the summer (July through September). Between October and March, they have been documented farther downstream to Princeton (RM 164) (Johnson et al. 1992, as cited in NMFS 1997). They probably are rearing throughout the Sacramento River at this time.

As fry and juvenile winter-run Chinook migrate downstream, they share the river with other juvenile salmonids (especially spring-run Chinook salmon) that spent the summer in cooler upstream areas. These other juvenile salmonids would be much larger than the more recently emerged winter-run fry; therefore, we hypothesize that differences in the size and age of juvenile salmonids from the various runs result in different habitat preferences, which helps to limit competition for resources.

Winters in the Central Valley of California are mild enough to allow for juvenile growth. Shasta Dam releases relatively warmer water in the winter, which also promotes winter growth. Many streams in the Pacific Northwest that have cool summer water temperatures tend to have winter temperatures that are too cold to allow for sufficient growth of juvenile salmonids. The low-temperature threshold for positive growth in juvenile Chinook salmon is believed to be about 40.1°F (4.5°C), with 39.4°F (4.1°C) being the lower limit for zero net growth in a juvenile Chinook population (Armour 1990, as cited in McCullough 1999). In the laboratory, juvenile sockeye salmon growth is greatly reduced at low temperatures even with full rations (Figure 4.2-4**).** At low temperatures juvenile Chinook salmon typically stop feeding and seek cover (McCullough 1999). In the Sacramento River, water temperatures rarely fall below 43°F (6°C), and in years when they do, they rarely last for more than a few days. Figure 4.2-5 shows the water temperature regime of the Sacramento River between 1970 and 2001, as modeled between Keswick Dam (RM 302) and RM 260 (Watercourse Engineering 2002). Figure 4.2-6 shows one of the few years (1972) in which water temperatures fell below 43°F (6°C) in the modeled reach, but these cold water temperatures lasted for only a few days in January. Water temperature data recorded farther downstream at Wilkins Slough (RM 118) also demonstrate that temperatures rarely fall below 43°F (6°C) in the lower Sacramento River (Figure 4.2-7). So the Sacramento

River provides ideal habitat for winter-run salmon: cool water temperatures that allow egg incubation in the summer in combination with warm winter rearing temperatures.

Juvenile salmonid growth is important because it influences survival. The duration of the winter-run salmon rearing period (5–10 months) is intermediate between that of fall-run Chinook (1–7 months [Yoshiyama et al. 1998]) and the yearling component of the spring run (12–15 months). Likewise, the smolts are of a size intermediate between fall-run smolts (2.8–3.3 in [70–85 mm]) and yearling spring-run smolts (5.9–7.9 in [150–200 mm]). Juvenile winter-run Chinook smolt between January and April at an average fork length of about 4.6 in (118 mm) (CDFG, unpublished data, as cited in NMFS 1997). The longer rearing period results in relatively large smolts and presumably higher survival during outmigration and early ocean rearing compared to the fall run (Bilton 1984, Martin and Wertheimer 1989, Unwin 1997, Myrick and Cech 2000). This relationship between larger size and higher survival is believed to be mostly a result of reduced vulnerability to predation. So, the winter-run strategy has the advantage of a stream-type life history without the summer juvenile rearing habitat limitations of the spring run or late-fall run.

4.2.5.2 Adaptation to the flow regime of the Sacramento River

For winter-run Chinook, access to cooler high-elevation reaches that make summer spawning and egg incubation feasible was likely difficult. There are often barriers to upstream migration of salmon in steeper streams (like the McCloud River) that can be ascended only during higher flows. Flows are typically much lower in May and June than earlier in the winter. Winter-run Chinook are known to migrate in winter and hold for 2–3 months prior to spawning, which is likely an adaptation for passing migration obstacles that may be impassable just prior to the winter-run spawning season when flows are lower.

Some researchers have noted that juvenile winter-run Chinook outmigrate in pulses that seem to coincide with high rainfall events accompanied by increased turbidity (Hood 1990, as cited in NMFS 1997). Smolt outmigration during winter or spring freshets likely reduces predation losses because turbidity can reduce the efficiency of visual predators such as piscivorous fish and birds, and increased outmigration rates reduce the amount of time fish are exposed to predators (Petersen and DeAngelis 1992, Berggren and Filardo 1993, Jager and Rose 2003). Freshets, especially in the spring, may further reduce predation by lowering water temperatures and reducing the feeding activity of warmwater predators such as largemouth bass (*Micropterus salmoides*) (Hathaway 1927).

4.2.5.3 Historical habitat limitations

Because eggs and alevins require cooler temperatures than any other salmonid life stage, winter-run Chinook are restricted to spawning in reaches that remain cold all summer. The areas where summer temperatures remained below the lethal threshold for eggs and alevins were spring-fed, higher-elevation streams such as the McCloud River (Figure 4.2-8). These reaches are steeper and more confined than reaches downstream in the Sacramento Valley and therefore had greater stream power. Consequently, the reaches historically used by winter-run Chinook for spawning were coarse-bedded—predominately cobble and boulder, with gravel typically occurring in patches where shear stress was controlled locally by flow obstructions or bank configuration (Figure 4.2-9). So winter-run Chinook were restricted not only in the linear extent of stream available to support spawning because of temperature constraints, but also in the amount of spawning gravel available to the population. Figure 4.2-10 shows spawning habitat in the

Sacramento River as a function of river mile, illustrating the dramatic differences in available spawning habitat between the upstream and alluvial reaches.

Extensive rearing habitat would have been available for winter-run Chinook under historical conditions because of their life history timing. Prior to the construction of large water supply dams in the basin, high flows between January and March often inundated extensive floodplains along the Sacramento River (Sommer et al. 2001b). These shallow-water, low-velocity areas tend to be highly productive, which confers bioenergetic advantages that promotes higher growth rates and, therefore, higher survival rates. The extent of historical flooding in the Sacramento River valley was vast (Kelley 1989), and winter-run salmon juveniles emerged and migrated downstream at a time that would have allowed them to take advantage of these prolonged periods of floodplain inundation.

4.2.5.4 Conceptual model of spawning habitat as a limiting factor

Most salmon with a stream-type rearing life history (e.g., coho salmon) have populations limited by availability of summer or winter rearing habitat. Winter-run Chinook are unique in that they have a stream-type rearing life history that is spawning-gravel-limited.

Limited availability of spawning gravels in the high-elevation reaches of the McCloud River, combined with extensive rearing habitat downstream, make it likely that competition for spawning habitat was, and still is, the primary source of density-dependent mortality for winter-run Chinook. These factors also suggest that winter-run juvenile production is well below the river's carrying capacity.

If there are no density-dependent population bottlenecks after spawning, then density-independent mortality after spawning would affect escapement size and year-to-year variability. Before construction of Shasta Dam, intragravel conditions were likely conducive to high egg-to-emergence survival (McBain 1989). Mortality due to predation, especially during fry and juvenile dispersal, may have been very high at times, but even small fall freshets would have provided opportunities for fry to disperse to areas with greatly reduced risk of predation. In addition to resident rainbow trout and age 1+ steelhead, bull trout—a highly piscivorous species—were also present, at least in the McCloud River, and probably throughout the Sacramento River in the winter. Because winter-run Chinook fry are very small in the fall and winter, a large proportion of the predator population could feed on them, including not only large predators like Sacramento pikeminnow, but also smaller trout and char.

4.2.6 Effects of anthropogenic changes on winter-run Chinook salmon habitat

Based on the above conceptual model, we hypothesize that the greatest potential threats to winter-run Chinook salmon are reductions in the quantity of spawning habitat either through increased temperatures or loss of gravels, and reduced opportunities for growth in winter. Other factors, such as reduced spawning gravel quality (e.g., changes in particle size distribution), increased risk of predation, and unscreened diversions may also adversely affect the population, but to a lesser degree.

The conceptual model sketched in Chapter 4.2.5 indicates that spawning gravel limitations are the most likely factor limiting the winter-run salmon population, and the most likely explanation for the dramatic decline in the winter-run salmon population beginning in the 1970s. Several pieces of evidence support this hypothesis, including:

1. the predilection for winter-run salmon to spawn in the most upstream reaches accessible to them, where the effects of bed coarsening since the completion of Shasta Dam are most pronounced;
2. evidence of redd superimposition in winter-run spawning areas, as indicated by aerial surveys;
3. estimates of the potential effects of superimposition on adult carrying capacity and egg mortality; and
4. the abrupt crash of the population in 1982, which is consistent with a severe increase in density-dependent mortality.

Each of these points is discussed in more detail below. To test the hypothesis that spawning gravel availability is limiting the winter-run salmon population, we analyzed several datasets, and developed and applied both a new sediment transport model (Cui, in press) and a new state-space winter-run salmon population model.

Chapter 4.2.5.3 suggests that rearing habitat for juvenile winter-run Chinook was likely extensive under historical conditions and there was relatively little competition from other juvenile salmonids for food and habitat. Juvenile winter-run also benefited from the frequent flooding that occurred in the Sacramento Valley, which provided enhanced growth opportunities on inundated floodplains in the lower Sacramento River during the winter months as juveniles migrated downstream to the Delta. Water supply development, flood management operations, and levee construction have reduced the frequency and duration of floodplain inundation in the Sacramento Valley, so winter-run juveniles currently enjoy few opportunities for floodplain rearing except when the bypasses are flooded.

4.2.6.1 Bed coarsening

As described in Section 4.2.2, the winter-run salmon population likely suffered at least one year-class failure in the years after the construction of Shasta Dam began (1943–1945). However, the population seemed to rebound by the 1950s and 1960s when escapements reached tens of thousands (Slater 1963). This initial increase in winter-run Chinook escapements was caused primarily by changes in the water temperature regime below Shasta Dam, which suddenly released the population from bottlenecks due to limited spawning habitat. Prior to the construction of Shasta Dam, Sacramento River water temperatures in the reaches below the Shasta Dam site were too warm to support spawning by winter-run Chinook, whereas temperatures in the McCloud River were suitable for spawning all the way to its confluence with the Sacramento River (Figure 4.2-11). Summer water temperatures below Keswick Dam are now much colder than they were historically (Figure 4.2-11). By releasing water from the hypolimnion, Shasta Dam provided suitable water temperatures in the alluvial reach of the Sacramento River, which had more gravel than the high-gradient reaches of the McCloud River that provided historical spawning habitat for winter-run salmon (Slater 1963). The increase in spawning habitat allowed the population to flourish by increasing juvenile production. We do not know how much gravel was available below Shasta Dam when it was first built; however, Figure 4.2-10 displays spawning habitat that was mapped downstream of Keswick Dam (RM 302) in 1964. The area of suitable spawning habitat may have been higher in the late 1940s and 1950s than that displayed in the map. However, subsequent changes to the quality of this spawning habitat led to corresponding declines in the winter-run population.

The gravel available downstream of the dam would be expected to decline over time because any dam that does not pass coarse sediment, yet continues to release flows capable of mobilizing the bed, will recruit gravel stored in the channel bed. As gravels are transported downstream by high flows, the bed becomes more coarse as large "lag particles," that are not mobilized by high flow releases cover a greater portion of the bed surface. Eventually the bed becomes composed of enough immobile particles (~50%) that bedload transport ceases. This condition is often referred to as an armored bed. Finer sediment, including spawning-sized gravel, is often trapped beneath the armor layer (see Figure 4.2-12). This phenomenon has been documented by numerous researchers (e.g., Stanley 1951, Livesay 1965, Hales et al. 1970, Pemberton 1976), and mathematically modeled by Ackers and White (1973). Williams and Wolman (1984) provide several examples of bed coarsening below dams (e.g., Figure 4.2-13). Bed coarsening first occurs in the reaches immediately below the dam because downstream reaches are still being supplied with gravel from upstream reaches until these too are depleted and armored. As a result, bed coarsening and armoring generally progress downstream over time (e.g., Figure 4.2-13).

How coarse the bed becomes is a function of channel morphology, slope, and flows released from the dam. Bed coarsening below dams may reach an equilibrium (i.e., remaining particles are immobile at the highest flow releases) at gravel sizes still suitable for salmon spawning (as in the Tuolumne River), or they may coarsen to the point that a female salmon can no longer move the substrate and build a redd (as in the Feather River [CDWR 2004]).

Shasta and Keswick dams effectively capture all coarse sediment transported from upstream reaches (Buer 1995). Compared to unimpaired flows, annual high flows in reaches downstream of the dams are now lower in some years, but higher in others (Figure 4.2-14). Many flows released since 1940 have been capable of transporting gravel. Bed coarsening is inevitable where sediment supply is reduced and gravel transporting flows continue to occur (Dietrich et al. 1989).

A spawning habitat survey conducted in 1987 provides evidence of bed coarsening below Keswick Dam. Finer sediment often occurs along channel margins because of the drag associated with banks, which reduce shear stress and therefore the capability of a given discharge to mobilize sediment. Figure 4.2-15 displays the location of winter-run redds in 1987, which are clustered along both channel margins, with no redds located in mid-channel. The distribution of spawning shown in 4.2-15 suggests that the channel bed has become too coarse in the center of the channel to allow spawning. In the reach between Keswick Dam (RM 302) and ACID Dam (RM 298.4), very little gravel was lost between 1964 and 1980 and the remaining gravel patches are all in protected areas (Figure 4.2-16). Aerial redd surveys in the 1990s continue to document redds in the locations mapped in the 1980 survey, which suggests that local hydraulics in these areas permit gravel to remain in storage even during high flow events.

Two other spawning habitat surveys conducted in 1964 and 1980 also provide evidence of bed coarsening in the upper Sacramento River. Between Keswick Dam (RM 302) and Anderson Bridge (RM 283), approximately 2,235,976 ft^2 (207,729 m^2) (54%) less spawning habitat area was mapped in 1980 than in 1964 (Figure 3-2), despite the addition of approximately 13,300 yds^3 (10,170 m^3) of spawning-size gravel within this reach in 1978 and 1979 (CDWR 1980). It should be noted that the 1964 and 1980 surveys were derived from observations of redd locations, so changes in spawning distribution and differences in escapements between the two surveys may have affected the amount and location of habitat mapped by the surveys. During the period between the surveys, there were changes in the system that may have affected upstream passage of adults (e.g., the barrier caused by Red Bluff Diversion Dam beginning in 1967), which may have affected the distribution of spawning. Escapements were higher in the mid-1960s as compared with the late 1970s, so the 1964 surveys may have mapped spawning habitat that was saturated, while the 1980 survey mapped only a portion of available spawning habitat because of a lower escapement. The habitat surveys also used different levels of resolution that may have affected the cumulative spawning area mapped, because the 1964 survey mapped general spawning areas, while the 1980 surveys mapped more specific patches of spawning habitat (CDWR 1980). Despite these complications, CDWR concluded that the changes in spawning habitat between the two surveys indicated a loss of habitat that reflected a trend of bed coarsening between ACID Dam (RM 298.4) and Anderson Bridge (RM 283) (CDWR 1980).

CDWR monitored the gravel that was added at Redding Riffle in 1979 (Figure 4.2-17), and the results reinforce the idea that since the closure of Shasta Dam, high flow events have scoured spawning-sized gravel from the upper Sacramento River, thereby reducing spawning habitat. After the gravel was placed at Redding Riffle, two high flow events of 36,000 cfs and 50,000 cfs occurred in the winter of 1980. Following these flow events, CDWR surveys of Redding Riffle indicated that nearly 85% of the placed gravel had been eroded from the injection site (CDWR 1980). CDWR concluded that discharges with magnitudes of 50,000 cfs could mobilize spawning-sized gravel from the reach of the upper Sacramento River below Keswick Dam. Since the completion of Shasta Dam in the mid-1940s, there have been several flow events with magnitudes greater than 50,000 cfs (Figure 3-1). These flow events likely scoured spawning size gravel from the channel bed, but the lack of sediment supply caused by Shasta Dam prevented the gravels from being replenished from upstream, thereby causing bed coarsening. Of the other enhancement sites shown in Figure 4.2-17, only Turtle Bay West was determined a success and that was because it was placed in a side channel with reduced shear stress (Parfitt and Buer 1981).

As a channel bed coarsens, the flows required to mobilize it increase, because all particles that can be mobilized will have already been transported downstream, thereby requiring ever higher flows to mobilize the coarser sediment particles. To assess the increasing discharge magnitudes required to mobilize a coarsening bed, we applied the new TUGS (The Unified Gravel Sand Model) sediment transport model (Cui, in press) to RM 294. There is no data available on the particle size distribution of the upper Sacramento River soon after Shasta Dam was completed, so we used professional judgment to assume an initial median grain size (D_{50}) of 2.51 in (63.7 mm) for this analysis of bed surface mobilization. The TUGS model predicts that the discharge at which mobilization occurs increased from 30,000 cfs in 1939 to 50,000 cfs after a large flood event in December 1939, to 80,000 cfs in 2004. Figure 4.2-18 shows predicted transport rates as a function of flow at different times since the construction of Shasta Dam.

We also applied the TUGS model to a reach of the upper Sacramento River (RM 295–RM 290) to test the degree to which the channel bed below Keswick Dam has coarsened since the completion of Shasta Dam. This reach was modeled because bed coarsening is likely to be most pronounced in this upstream reach because of its proximity to Keswick Dam (RM 302) and its location above Clear Creek (RM 290), which is the first significant sediment source to the mainstem Sacramento River channel. This reach is also important as a spawning area for multiple runs of Chinook salmon, including the endangered winter-run Chinook salmon. Figure 4.2-19 displays the thalweg profile and the location of cross-section surveys that were used as TUGS model input to represent channel geometry and slope. The higher number of cross section surveys available for the reach between RM 295 and RM 290 (Figure 4.2-19) also contributed to its selection as the test reach from among the three candidate reaches.

The TUGS model can be used to predict particle size distribution of both the bed surface and subsurface, including the percentage of fine sediment stored in a channel bed. However, for this analysis, we focused on predicting the median grain size mobilized by different flows at each of the cross sections in the reach between RM 295 and RM 290. The D_{50} predicted by the TUGS model is a cross-sectional average; however, in the stream, grain size distribution is considerably heterogenous, with some areas being coarser and others finer. Therefore, the TUGS model results from this analysis are most useful for assessing overall trends in channel bed composition and in-channel gravel storage, rather than for predicting changes in spawning habitat.

Figure 4.2-20 shows modeled changes in gravel storage in Reach 3, suggesting that there was a cumulative loss of 75,861 yds^3 (58,000 m^3) in the modeled reach between the initiation of Shasta Dam construction and 1990. These results suggest that extremely high flow events (e.g., peak flow of 186,000 cfs in 1940, as measured at the USGS Keswick gauge [No. 11370500]) play a large role in changing the amount of sediment stored in the channel bed. The modeling results also suggest that gravel augmentation implemented since 1990 (~230,000 yds^3 [175,848 m^3]) in reaches downstream of Keswick Dam has since partially compensated for the loss of gravels in this reach.

We also applied the TUGS model to predict changes in median grain size within the modeled reach as a way of assessing the degree of bed coarsening that has occurred since the completion of Shasta Dam. The results suggest that high flow events like the one in 1940 can cause rapid change in the grain size composition of the channel bed (Figure 4.2-21). Successive changes in grain size distribution may be more gradual, in large measure because of the increasing discharge magnitudes required to mobilize the coarser bed. The predicted median grain size for the year 2005 (Figure 4.2-21) ranges from 3.1–4.3 in (80–110 mm) at the four locations. The D_{50} at RM 290.1 reached approximately 3.9 in (95 mm) by 1990; however, gravel augmentation resulted in re-fining of the bed. At the four sites where gravel augmentation did not occur, the median grain size is predicted to have reached a size that only the largest salmon can use for spawning (e.g., the D_{50} of the largest gravels that a 35.4-in (90-cm) salmon would successfully build a redd in is approximately 3.7 in [95 mm]). Recent carcass surveys indicate that the mean fork length of female winter-run Chinook salmon is 29.1 in (74 cm) (Snider et al. 1998a), which suggests that most winter-run Chinook salmon would not be able to use gravels at the modeled sites. As shown in Figure 4.2-21, the TUGS model predicts that the bed has continued coarsening through the 1990s between RM 295 and RM 291.6, though the bed may be reaching equilibrium. To help test the bed coarsening hypothesis, Stillwater Sciences conducted an aerial spawning habitat survey in 2005, which will be compared with those from previous years (1964 and 1980) to determine the extent to which gravels continue to be lost, or the extent to which recent gravel augmentation has succeeded in ameliorating bed coarsening in the upper Sacramento River. The results of this analysis will be included in the final project report of the Sacramento River Ecological Flows Study, which is currently scheduled for distribution in the fall of 2007.

We conducted a sensitivity analysis to assess the degree to which the TUGS model results were sensitive to assumptions about initial grain size. Table 4.2-3 and Figure 4-2-22 suggest that the degree to which the bed coarsens is relatively insensitive to initial bed size.

Table 4.2-3. Results from 2005 modeling of D_{50} at four locations in the Sacramento River, assuming an initial D_{50} of 2.51 and 3.09 in (63.7 and 78.4 mm) in 1940.

Sacramento River Mile	Modeled 2004 D_{50} values			
	Initial D_{50} of 2.51 in (63.7 mm) in 1940 (Figure 4.2-21)		Initial D_{50} of 3.09 in (78.4 mm) in 1940 (Figure 4.2-22)	
	in	mm	in	mm
294.0	4.17	106.0	3.97	100.8
295.0	3.80	96.9	3.68	93.4
291.6	3.71	94.2	3.64	92.5
290.1	3.44	87.3	3.39	86.0

4.2.6.2 Restricted spawning distribution and redd superimposition

Winter-run Chinook salmon appear to select spawning areas in the Sacramento River based on their location in the watershed, rather than on gravel suitability or availability. Examination of Figure 4.2-1 indicates that in 1985, a significant fraction of winter-run Chinook spawned below RBDD (RM 243.5). In 1986, the gates at RBDD were raised during the winter to allow winter-run Chinook salmon to pass. Since 1990, relatively little winter-run spawning has occurred downstream of the RBDD. This pattern was repeated in 2001 when fish passage facilities were improved at ACID Dam (RM 298.5), after which very little spawning occurred downstream, despite the greater availability of spawning gravels downstream as mapped by the 1980 survey (Figure 4.2-23).

A CDFG report dated 13 August 1981 highlights the potential disconnect between escapements and available spawning habitat (CDFG 1981):

> We are becoming increasingly concerned about the future of winter-run chinook which, so far as we know, are unique to the Sacramento River. We suspect that a three-year cycle is the normal pattern for winter-run instead of the usual three to four-year cycle for fall-run. Returning winter-run adults from the two drought years were extremely low (1–2 thousand). This leaves only one strong year class out of the three-year cycle. That strong year class is completing spawning now.
>
> The last time we talked to Dick Hallock he indicated an estimated 19,000 winter-run had passed Red Bluff Diversion Dam this year. From our flights of the river on July 17 and August 4, we are at a loss to explain where the 19,000 salmon spawned.

Quantitative aerial redd surveys were conducted in June 1981 (prior to the flights described above). On June 11, 30 redds were counted at RM 296, about 6 miles below Keswick Dam, and 3 at the “Osborne Riffle” (RM 241) downstream of Red Bluff Diversion Dam. On June 24, 79 new redds were counted in the 17-mi reach below Keswick Dam, 60 of which were at the RM 296 gravel patch, and 11 below Red Bluff Diversion Dam (Figures 4.2-24 and 4.2-25). At the peak of the winter-run spawning season, with an escapement of 19,000 fish (approximately 9,500 females) past RBDD, only 79 redds were counted, with a total of only 113 redds counted to date. Ninety of these 113 redds were constructed in a single gravel patch.

4.2.6.3 Population dynamics modeling to test hypotheses regarding changes in spawning habitat

The dramatic decline in the winter-run Chinook salmon population from the 1970s to the 1980s strongly suggests an abrupt reduction in the carrying capacity of its habitat (i.e., an increase in density-dependent mortality). The apparent recovery of the population in 1981 (Figure 4.2-2) at first seems inconsistent with this conclusion, but additional analysis suggests that it may actually confirm it. The low escapements in 1979 and 1980 were due to unusually high water temperatures released at Shasta Dam during the winter-run Chinook salmon incubation period in 1976 and 1977 (Figure 4.2-26) (Hallock and Fisher 1985). The elevated water temperatures were a source of density-independent mortality. The change in density-dependent mortality would have had to occur after 1978 and before 1981, as the 1978 spawners produced the large escapement of 1981 (most winter-run return at age 3). Under this hypothesis, the 19,000 adult fish returning in 1981 would have encountered greatly reduced spawning habitat, which resulted in the very low escapement of 1984.

We developed and applied a state-space population model to test the hypothesis that a reduction in spawning habitat between 1978 and 1981 could explain winter-run population dynamics. State-space modeling is a technique for incorporating stochastic variability into more mechanistic models of temporal processes, accounting in a rigorous way for autocorrelation and observation error (Buckland et al. 2004). In modern formulations, it is extremely general; it can be used with non-linear processes (such as compositions of stock-production functions), and with non-Gaussian process and observation errors. A state-space model is a statistical model, associated with a calculable likelihood function, and as such allows conventional statistical tools, such as maximum-likelihood estimation and Bayesian methods, to be applied to questions involving model parameters. This approach allows for stochastic variability to be incorporated, as well as functions that relate population parameters to environmental variables.

A high-flow event in February 1980 was capable of transporting significant amounts of sediment because of its magnitude (peak flow of 51,300 cfs) and duration (7 days with mean daily flows exceeding 50,000 cfs), as measured at the Keswick Dam gauge (USGS no. 11370500). Parfitt and Buer (1981) described the effects of those flows on recently emplaced gravel at the Redding Riffle Site (RM 298) (Figure 4.2-17): "In total, about 85 percent of the area spread with the imported gravel degraded to or below the level of the channel prior to its placement, and a like percentage of the total volume was removed by the high water flows."

Figure 4.2-27 shows the results of a state-space model incorporating water temperature during incubation, harvest (ocean and in-river), and a change in density-dependent mortality in February 1980 and after 1998 gravel augmentation. The model uses a Beverton-Holt stock-recruitment relationship to represent density-dependent mortality and estimates a pre-1980 carrying capacity of 900 and a post-1980 carrying capacity of 12,500 (Figure 4.2-28). The model is able to capture the trends in population size from 1974 to the present (Figure 4.2-27).

4.2.6.4 Increased fine sediment

As discussed in section 4.2.3.2, the percentage of fine sediment (< 0.08 in [< 2 mm] diameter) in the channel bed is an important regulator of salmonid survival from egg incubation to fry emergence (McCuddin 1977, Reiser and White 1988). In some spawning areas, detrimental concentrations of fine sediment result from locally high sediment supply, which may suffocate incubating eggs and entomb emerging fry. However, this is not the case for winter-run Chinook salmon on the upper Sacramento River. Sediment supply is low, because (1) upstream dams block the transport of sediment to these reaches, (2) bank erosion is minimal, and (3) there are few significant sediment sources to the mainstem Sacramento River between Keswick Dam (RM 302) and the mouth of Clear Creek (RM 290). Bulk sampling completed by CDWR in 1995 in the upper Sacramento River suggest that fine sediment concentrations in spawning areas are relatively low (CDWR 1995). Nevertheless, the possibility that fine sediment concentrations in the bed have (or may in the future) become detrimentally high in key winter-run Chinook spawning reaches cannot be entirely ruled out. Even small additions of fine sediment can, over time, accumulate to detrimental levels if they are not periodically flushed out by sufficiently high, bed-mobilizing flows. In addition, sand and fine sediment from eroding banks and agricultural runoff can often be transported as bedload by baseflows, which may cause ongoing infiltration of fine sediment into the matrix of coarser sediment that serves as a framework for the bed.

Bed-mobilizing flows in the winter-run spawning reaches of the upper Sacramento River occur much less frequently than before the dams were in place, due to reductions in the magnitude of peak winter flows and coarsening of the bed surface. As described above, an application of the TUGS model at RM 294 indicates that bed coarsening increased the bed mobilization flow from 30,000 cfs in 1939, to 50,000 cfs in 1940, to 80,000 cfs in 2004.

If survival to emergence is being reduced by fine sediment, it would have important implications for the winter-run population, because greater than expected mortality in these earliest life stages would be expected to propagate in direct proportion to reduced adult escapements (i.e., any increase in density-independent mortality after the spawning bottleneck will result in a decrease in returning adults). Moreover, even if spawning habitat quality has not yet been adversely affected by fine sediment, it may still be in jeopardy, because fine sediment concentrations will presumably continue to increase unless the frequency of bed-mobilizing flows is restored. This highlights the critical importance of confirming whether fine sediment is affecting winter-run spawning habitat.

4.2.7 Potential management alternatives

Based on the conceptual model, field data, and analyses using sediment transport, state-space, and stock-recruitment modeling, and based on the anthropogenic impacts described above, management actions designed to reduce superimposition are most likely to succeed at increasing escapement of winter-run Chinook salmon in the Sacramento River. Reductions in superimposition could be achieved through two ways: increasing spawning area, or increasing spawning distribution. Because the evidence suggests that gravel availability is the primary limiting factor, management of the water temperature compliance point or increasing mainstem rearing habitat would not be expected to have as great an effect on winter-run numbers, as described below.

4.2.7.1 Gravel augmentation to increase spawning habitat downstream of Keswick Dam

Although population modeling indicates that superimposition is a factor currently limiting escapement, a superimposition field study should be conducted to test model results before any management actions are implemented. Long-term superimposition monitoring is recommended to ensure the effectiveness of any management actions taken.

Gravel augmentation below Keswick Dam is recommended to increase spawning habitat for the winter run and decrease superimposition. Although most spawning currently occurs below Keswick Dam, relatively little gravel is available there. Gravel augmentation is expected to allow more females to spawn, and thus increase production (Figure 4.2-29). ESCAPE model results indicate that under current conditions, escapements of approximately 1,500 females saturate available spawning habitat, such that additional spawners fail to increase production because of egg mortality associated with redd superimposition.

4.2.7.2 Managing passage at ACID Dam to increase spawning distribution

In addition to gravel augmentation, the migration of winter-run Chinook salmon past ACID Dam could be managed to better distribute spawners upstream and downstream. Figure 4.2-1 illustrates that prior to improving passage at ACID Dam in 2001, many winter-run Chinook salmon spawned successfully in the habitat between Airport Road and ACID Dam. By distributing spawners between habitat upstream and downstream of ACID Dam, more females should be able to spawn successfully, and production should increase, especially if combined with gravel augmentation downstream of Keswick Dam. ESCAPE model results indicate that by controlling migration past the ACID Dam and adding gravel downstream of Keswick Dam, over 10,000 females could spawn before egg mortality due to superimposition would begin to significantly limit production.

Effective management of passage at ACID Dam would require determining the appropriate numbers of migrants to allow past the dam to spawn upstream each year. However, the number of spawners allowed to pass should depend on the escapement. For example, if only 2,000 females return, less than 1,000 would be allowed to pass upstream so that egg mortality from superimposition is negligible. If 8,000 females return, then over 2,000 should be allowed to pass upstream so that all gravels are saturated, even though superimposition rates will be higher. Superimposition monitoring is recommended to test model results and to support adaptive management of passage targets.

In addition to monitoring the response of the winter-run salmon population to gate closure, this action should also include monitoring to assess the potential effects of denying passage to other migratory aquatic species.

4.2.7.3 Water temperature compliance point

Recent USBR management of the Shasta and Trinity divisions of the CVP have been successful in providing suitable water temperatures throughout the primary spawning reach of winter-run salmon (RM 302–RM 284), while maintaining sufficient coldwater storage to support spawning in the successive year to hedge against drought conditions. The recent upstream shift in winter-run spawning (Figure 4.2-1), coupled with the operation of the Shasta Temperature Control Device (TCD) may also facilitate water temperature management for winter-run salmon during dryer years. During such periods, the Sacramento River Water Temperature Task Group has the option of contracting the coldwater zone upstream to protect the vast majority of constructed redds, thereby maintaining reservoir carryover storage to preserve the coldwater pool and allow for water temperature management in the following year.

The water temperature compliance point fluctuates each year, but it is typically located downstream of Balls Ferry (RM 276). Very few winter-run Chinook spawn downstream of Airport Road (RM 284), so moving the water temperature compliance point from Bend Bridge (RM 260) to Ball's Ferry (RM 276) would likely have little effect on the success of winter-run spawning and egg incubation. However, there is little information about where winter-run adults hold in the Sacramento River, so it is not clear if a contraction of the cold water zone would increase water temperatures in key areas of winter-run holding. It is likely that winter-run adults hold in habitats near where they spawn, which would suggest that winter-run adults generally hold in areas between Keswick Dam (RM 302) and a location near Airport Road Bridge (RM 284). Consequently, moving the temperature compliance point upstream to Ball's Ferry (RM 276) would likely have little effect on winter-run holding habitat and egg viability within holding females. Nevertheless, it would be prudent to survey the upper Sacramento River to identify the primary holding areas for winter-run salmon before contracting the cold water zone.

Moving the water temperature compliance point upstream could also affect winter-run fry rearing habitat, thereby influencing fry growth rates, and survival. The majority of winter-run fry begin dispersing downstream soon after emergence. As they grow, they become more tolerant of higher water temperatures, and if food resources are abundant, higher water temperatures can promote faster growth, which typically increases survival. However, higher water temperatures can also stress salmonid fry and make them vulnerable to other factors affecting health and survival, especially in the absence of an abundant food supply. There have been a few studies of invertebrate production (Stillwater Sciences 2003, USFWS 2005a, CFDG 1983), juvenile salmonid growth (Limm and Marchetti 2003), and salmonid rearing habitat in the Sacramento River (CDFG 1997, 1998, 1999, 2000; USFWS 2005b). However, it is not clear how the upstream movement of the water temperature compliance point would likely affect salmonid rearing habitat conditions in the upper Sacramento River. Rearing habitat conditions in the mainstem channel are especially important for winter-run salmon fry, because they rarely benefit from any floodplain or bypass flooding because of their emigration timing. In contrast, the juveniles of other salmonid runs periodically benefit from floodplain and bypass flooding, which can promote faster growth and higher survival (Sommer et al. 2001) and contribute to strong year classes.

Moving the water temperature compliance point upstream to Ball's Ferry (RM 276) could also affect winter-run fry survival by influencing the distribution and abundance of potential predators. Colder water temperatures can deter centrarchids from migrating farther upstream, and it can also depress predator feeding activity. Contraction of the cold water zone in the Sacramento River could allow predators to move farther upstream in higher densities, thereby increasing the predation exposure of winter-run fry. Little is known about the distribution and abundance of potential salmonid predators in the Sacramento River and how water temperatures can influence predator distribution, abundance, and feeding activity. Consequently, it is difficult to predict if movement of the water temperature compliance point would significantly affect the predation mortality of winter-run fry. As described in Chapter 4.2, fry production is especially critical for winter-run Chinook salmon, so it seems prudent to research the effects of water temperature on predator distribution, abundance and survival before contracting the cold water zone in the Sacramento River.

4.3 Spring-run Chinook Salmon

Spring-run Chinook salmon were probably the most abundant salmonid in the Central Valley under historical conditions (Mills and Fisher 1994), but large dams eliminated access to vast amounts of historical habitat and the spring run has exhibited the severest declines of any of the four Chinook runs in the Sacramento River basin (Fisher 1994). Dams may also have reduced or eliminated spatial segregation between spawning spring- and fall-run Chinook in some areas, particularly in the mainstem Sacramento River, leading to increased potential for hybridization on the spawning grounds.

The majority of spring-run Chinook used to spawn upstream in tributaries rather than the mainstem Sacramento River; however, the completion and operation of Shasta Dam made water temperatures suitable in the main stem downstream of Keswick Dam, which permitted spring-run salmon to spawn there. Because of hybridization with fall-run Chinook in the mainstem channel, there are only three "pure" populations of spring-run salmon remaining in Deer, Mill, and Butte creeks. Battle Creek provides one of the few opportunities for establishing a significant new population of spring-run salmon in the upper Sacramento River basin.

The Central Valley spring-run Chinook salmon ESU was federally listed as threatened on 16 September 1999 (NMFS 1999a). The threatened status of Central Valley spring-run Chinook salmon was reaffirmed in NMFS's final listing determination issued on 28 June 2005 (NMFS 2005a). Critical habitat for Central Valley spring-run Chinook salmon was designated by NMFS on September 2, 2005 (NMFS 2005b).

4.3.1 Distribution

Spring-run Chinook salmon once occupied all major river systems in California where there was access to cool reaches that would support oversummering adults. Historically, they were widely distributed in streams of the Sacramento-San Joaquin basin, spawning and rearing over extensive areas in the upper and middle reaches (elevations ranging 1,400–5,200 ft [450–1,600 m]) of the San Joaquin, American, Yuba, Feather, Sacramento, McCloud, and Pit rivers (Myers et al. 1998) (Figure 4.2-8). Only two ESUs of spring-run Chinook salmon remain in California: a Sacramento-San Joaquin population and a Klamath-Trinity population (Moyle et al. 1995). Spring Chinook runs in the San Joaquin River were extirpated in the mid- to late 1940s following the closure of Friant Dam and diversion of water for agricultural purposes to the San Joaquin Valley. In the Sacramento River, the closure of Shasta Dam in 1945 cut off access to the spring run's major historical spawning grounds in the McCloud, Pit, and upper Sacramento rivers. This represented a loss of 70% of spring-run spawning habitat in the Sacramento River basin (Yoshiyama et al. 2001). Populations of spawning spring-run Chinook in the Sacramento River basin are more common in east-side tributaries to the Sacramento River upstream of the mouth of the American River. The most important spawning populations are in Deer, Mill, and Butte creeks, because of their relative lack of past hatchery influence, as well as their relatively stable numbers. Some spawning also takes place in Big Chico, Antelope, Cottonwood, Beegum, Clear, and Battle Creeks, and the mainstem Sacramento River downstream of Keswick Dam and upstream of Red Bluff Diversion Dam (Cramer and Demko 1997, CDFG 1998, as cited in NMFS 2004b, CDFG 2002b, CDFG 2005 [GrandTab spawning data]). A spring run in the Feather River basin is maintained by hatchery production, but the stock is believed to have been hybridized with the fall run to a great extent (Lindley et al. 2004).

4.3.2 Population trends

At one time, spring-run Chinook salmon may have been the most abundant race in the Central Valley, with escapement in the hundreds of thousands (Mills and Fisher 1994). Spring-run Chinook salmon have since declined to remnant populations totaling a few thousand fish, sometimes approaching 30,000 to 40,000 in good years (Mills and Fisher 1994, NMFS 1999a). Loss of access to upstream spawning and rearing areas due to the construction of dams in the Sacramento and San Joaquin rivers is believed to have been a major cause of the decline of the spring run.

Under historical conditions, it is doubtful that spring-run Chinook salmon spawned in the mainstem Sacramento in significant numbers (Lindley et al. 2004). After the closure of Shasta and Keswick dams, spring Chinook salmon began to spawn in the mainstem Sacramento River when changes in temperatures made this a viable life-history strategy. Figure 4.3-1 displays annual escapements of spring-run Chinook salmon that spawn in the mainstem Sacramento River. Throughout the 1970s and 1980s, thousands of spring-run Chinook passed Red Bluff Diversion Dam (RBDD) en route to spawning grounds farther upstream. By the 1990s, escapements had declined; however, changes in the RBDD gate operations beginning in 1986 complicated the process of estimating spring-run Chinook abundance. Identification of the spring run at RBDD is also complicated by their low escapements and the difficulty of distinguishing fish of this run from those of the fall run. The two runs cannot be distinguished reliably by physical characteristics or run timing (Healey 1991), due to the naturally protracted run timing of the abundant fall run, and the apparent shift to later upstream migration timing by the spring run, which results in the runs being more temporally overlapped than they were historically.

Populations of spring-run Chinook salmon in Deer, Mill, and Butte creeks have been increasing since the 1990s (NMFS 2003). Butte Creek currently has the largest naturally spawning spring-run population. A few naturally spawning fish are also present in Battle, Clear, Cottonwood, and Big Chico creeks (CDFG 2005 GrandTab spawning data). In general, spring-run Chinook that are most genetically similar to the runs that occurred historically in the Sacramento basin are currently confined to spawning primarily in Deer, Mill, and Butte creeks, with perhaps a few spawning in the mainstem Sacramento River.

4.3.3 Life history

Spring-run Chinook display a stream-type life history strategy—adults migrate upstream while sexually immature, hold in deep cold pools over the summer, and spawn in late summer and early fall. Juvenile outmigration is highly variable, with some juveniles outmigrating in winter and spring, but others oversummering and then emigrating as yearlings. Table 4.3-1 illustrates life history timing for spring-run Chinook salmon in the Sacramento River basin. The table illustrates some of the changes in timing that have been observed for the run over the years, particularly with regard to upstream migration and spawning.

Table 4.3-1. Life history timing of spring-run Chinook salmon in the Sacramento River basin.

Life stage	Month																							
	Jan		Feb		Mar		Apr		May		Jun		Jul		Aug		Sept		Oct		Nov		Dec	
Adult entry into Sacramento-San Joaquin Delta Estuary	░	░	░	░	░	░	░	░																
"Historical" adult migration past Red Bluff Diversion Dam[1]							░	▓	▓	▓	░	░												
"Current" adult migration past Red Bluff Diversion Dam[2]					?	?	?	░	░	░	░	░	░	▓	░	░								
Entry into spawning tributaries (current)[3]				░	░	░	░	░	▓	▓	░	░												
Adult holding							░	░	░	░	░	░	░	░										
Historical spawning in Sacramento River basin[4]															░	░	░	░	░	░				
Spawning (Deer, Mill, Butte creeks[5])																░	▓	▓	▓	░				
Spawning (mainstem Sacramento River[6])																			░	░	░			
Incubation	░	░	░	░	░	░										░	░	░	░	░	░	░	░	░
Fry emergence	░	░	░	░	░	░															░	░	░	░
Fry/juvenile outmigration from tributaries[7]	░	░	░	░	░	░	░	░	░	░	░	░									░	░	░	░
Subyearling/Yearling outmigration from tributaries[7, 8]	▓	▓	░	░	░	░													░	░	▓	▓	▓	▓
Ocean entry (yearlings)	░	░	░	░	░	░	░	░													░	░	░	░

Sources: Fisher 1994, Meyers et al. 1998, Hill and Weber 1999, Ward and Reynolds 2001, C. Harvey pers. comm. 2003, USFWS AFRP 2005

[1] As observed in the 1970s (Cramer and Demko 1997).

[2] As observed in the 1980s (Cramer and Demko 1997).

[3] C Harvey (CDFG, Redding, pers. comm.., as cited in Cramer and Demko 1997); Hill and Webber (1999)

[4] Rutter (1908), Parker and Hanson (1944).

[5] Harvey (1995, as cited in Cramer and Demko 1997); Moyle (pers. obs., as cited in Moyle et al. 1995)

[6] F. Fisher (pers. comm., CDFG, Red Bluff, as cited in Cramer and Demko 1997).

[7] Some spring-run disperse downstream soon after emergence as fry in March and April, with others smolting after several months of rearing, and still others remaining to oversummer and emigrate as yearlings (USFWS 1995, as cited in Yoshiyama et al. 1998).

[8] Based on outmigrant trapping in Butte Creek in 1999 and 2000, up to 69% of age 0+ juveniles outmigrate through the lower Sacramento River and Sacramento-San Joaquin Delta between mid-November and mid-February, with a peak in December and January (CDFG 1998, Hill and Weber 1999, Ward and Reynolds 2001). A smaller number remain in Butte Creek and outmigrate in late spring or early summer, and in both Butte and Mill creeks, some of these oversummer and outmigrate as yearlings from October to March, with a peak in November (S.P. Cramer and Associates 1997, Hill and Webber 1999).

░	Period of activity
▓	Period of peak activity

4.3.3.1 Adult upstream migration and spawning

Age of return

Adult spring-run Chinook salmon may return between the ages of 2 to 5 years. Historically, adults of this run are believed to have returned predominantly at ages 4 and 5 years at a large size. Most spring Chinook salmon now return at age 3, although some portion returns at age 4 (Fisher 1994, McReynolds et al. 2005), probably due to intense ocean harvest (which removes the largest fish from the population and selects for fish that spend fewer years at sea). In 2003, an estimated 69% of the spring run in Butte Creek returned at age 4 (Ward et al. 2004); however, in most years, the proportion of age 4 adults is much smaller.

Timing of upstream migration

Adult spring-run Chinook salmon enter the Sacramento-San Joaquin Delta beginning in January, entering their natal spawning streams from March to July (Myers et al. 1998). Adults enter Deer and Mill creeks beginning in March, peaking in May, and concluding in June (Vogel 1987a, b; C. Harvey, pers. comm., CDFG, Redding; as cited in Cramer and Demko 1997). Their upstream migration is timed to take advantage of spring snowmelt flows, which allow them access to upstream holding areas where temperatures are cool enough to hold over the summer prior to the spawning season (NMFS 1999a). In the Sacramento River, upstream migration of spring-run Chinook overlaps to a certain extent with that of winter-run Chinook (December through July, peaking in March), and adults from particular runs are not generally distinguishable from one another by physical appearance alone, making it difficult to pinpoint migration timing with precision (Healey 1991).

Timing of spawning

The timing of spring run spawning in the mainstem Sacramento River has shifted later in the year, which is believed to be a result of genetic introgression with the fall run (Cramer and Demko 1997). Populations in Deer and Mill creeks, which do not appear to have significantly hybridized with the fall run, generally spawn earlier than those in the main stem (Lindley et al. 2004). Rutter (1908) noted that most spawning in the late 1800s/early 1900s in the Sacramento River basin occurred in August. Parker and Hanson (1944) observed intensive spawning of spring-run Chinook from the first week of September through the end of October in 1941. Currently, redd counts indicate that spring-run chinook spawning typically begins in late August, peaks in September, and concludes in October in both Deer and Mill creeks (Harvey 1995, as cited in Cramer and Demko 1997; Moyle, pers. obs., as cited in Moyle et al. 1995; NMFS 2004b).

Important holding and spawning areas

In Sacramento River tributaries, adults will pack densely in the limited holding pool habitat that is available. Some fish remain to spawn at the tails of the holding pools, while most move upstream to the upper watersheds to spawn, and still others move back downstream to spawn. Though there are several deep pools in the upper Sacramento River that may provide holding habitat for adult spring-run Chinook salmon, it is not clear which pools are heavily used. In Deer Creek, spring-run Chinook hold and spawn primarily in the 30 miles between the Ponderosa Way Bridge (elevation 1,640 ft) and upper Deer Creek falls (3,600 ft), which is apparently a barrier to further upstream movement (Marcotte 1984, Harvey 1994). The reach from the Ponderosa Way Bridge to the lower Highway 32 bridge crossing has been identified as important for summer holding (P. Moyle, pers. comm., as cited in Cramer and Demko 1997). In Mill Creek, spring-run spawning has been observed over 50 miles of stream from near the boundary of Lassen National Park at an elevation of 5,000 ft, downstream to the confluence of Little Mill Creek at an elevation of 800 ft

(Harvey 1994, as cited in Cramer and Demko 1997). Spawning habitat in Butte Creek is confined to lower elevations than in Deer and Mill creeks, with the highest densities of fish spawning in the approximately 10 miles between the upper limit to migration at Quartz Bowl, located approximately one mile below Centerville Head Dam (elevation 1,130 ft) downstream to Covered Bridge (elevation 400 ft) (Cramer and Demko 1997).

4.3.3.2 Egg incubation and alevin development

In the Sacramento River and its tributaries, egg incubation for spring-run Chinook extends from August to March (Fisher 1994, Ward and McReynolds 2001). Egg incubation generally lasts between 40 and 90 days at water temperatures of 42.8 to 53.6°F (6 to 12°C) (Vernier 1969, Bams 1970, Heming 1982, all as cited in Bjornn and Reiser 1991). At temperatures of 37°F (2.7°C), time to 50% hatching can take up to 159 days (Alderdice and Velsen 1978, as cited by Healey 1991). Alevins remain in the gravel for two to three weeks after hatching while absorbing their yolk sacs. Emergence from the gravels occurs from November to March in the Sacramento River basin (Fisher 1994, Ward and McReynolds 2001).

4.3.3.3 Juvenile rearing and outmigration

Fry and juvenile rearing takes place in the natal streams, the mainstem of the Sacramento River, inundated floodplains (including the Sutter and Yolo bypasses), and the Delta. During the winter, some spring-run juveniles have been found rearing in the lower portions of non-natal tributaries and intermittent streams (Maslin et al. 1997, Snider et al. 2001).

The rearing and outmigration patterns exhibited by spring-run Chinook salmon are highly variable, with fish rearing anywhere from 3 to 15 months before outmigrating to the ocean (Fisher 1994). Variation in length of juvenile residence may be observed both within and among streams (e.g., Butte versus Mill creeks, USFWS 1995, as cited in Yoshiyama et al. 1998). Some may disperse downstream soon after emergence as fry in March and April, with others smolting after several months of rearing, and still others remaining to oversummer and emigrate as yearlings (USFWS 1995, as cited in Yoshiyama et al. 1998). Scale analysis indicates that most returning adults have emigrated as subyearlings (Myers et al. 1998). Calkins et al. (1940, as cited in Myers et al. 1998) conducted an analysis of scales of returning adults and estimated that greater than 90% had emigrated as subyearlings, at about 3.5 in (88 mm).

The term “yearling” is generally applied to any juveniles that remain to oversummer in their natal stream. Yearling outmigrants are common in Deer and Mill creeks, but rare in Butte Creek (Cramer and Demko 1997). Extensive outmigrant trapping in Butte Creek has shown that spring-run Chinook emigrate primarily as juvenile (age 0+) fish from November through June, with a small proportion remaining to emigrate as yearlings beginning in mid-September and extending through March, with a peak in November (S. P. Cramer and Associates 1997, Hill and Webber 1999, Ward et al. 2004).

Coded-wire-tag studies conducted on Butte Creek spring-run Chinook have shown that juveniles use the Sutter Bypass as a rearing area until it begins to drain in the late winter or spring (Hill and Webber 1999). Few juvenile Chinook are observed in the bypass after mid-May. Five recaptures indicate that juveniles leaving the Sutter Bypass migrate downstream rapidly and do not use the mainstem Sacramento River as rearing habitat (Hill and Webber 1999).

Very little information is available on the estuarine rearing of spring-run Chinook (NMFS 2004b). NMFS (2004b) postulates that, because spring-run Chinook yearling outmigrants are larger than fall-run Chinook smolts, and ready to smolt upon entering the Delta, they may spend little time rearing in the estuary. Most have presumably left the estuary by mid-May (CDFG, unpublished data). Once in the ocean, spring-run Chinook salmon perform extensive offshore migrations before returning to their natal streams to spawn.

4.3.4 Specific habitat requirements

General habitat requirements for Chinook salmon are described in Section 4.1. Only habitat requirements specific to spring-run Chinook salmon are described here.

4.3.4.1 Holding habitat

Adult spring-run Chinook require large, deep pools with moderate flows for holding over the summer prior to spawning in the fall. Marcotte (1984) reported that suitability of pools declines at depths less than 7.9 ft (2.4 m) and that optimal water velocities range from 0.5 to 1.2 ft/s (15 to 37 cm/s). In the John Day River, Oregon, spring-run adults usually hold in pools deeper than 4.9 ft (1.5 m) that contain cover from undercut banks, overhanging vegetation, boulders, or woody debris (Lindsay et al. 1986).

Water temperatures for adult spring Chinook holding and spawning are reportedly best when < 60.8°F (16°C), and lethal when > 80.6°F (27°C) (Hinze 1959, Boles 1988, CDFG 1998). Spring Chinook in the Sacramento River typically hold in pools below 69.8–77°F (21–25°C). Adults may be particularly sensitive to temperatures during July and August, when energy reserves are low and they are preparing to spawn.

Butte Creek water temperatures have historically exceeded ideal temperatures for holding and spawning spring Chinook. There is evidence that spring-run in the San Joaquin River were also exposed to high temperatures during migration and holding (Clark 1943, Yoshiyama et al. 2001). It is possible that Central Valley spring Chinook are adapted to tolerate warmer temperatures than other Chinook stocks, but there is no experimental evidence to confirm this hypothesis, and short-term exposure to temperatures as high as 25–27°C (77–80.6°F) are known to be tolerated by adult Chinook salmon (Piper et al. 1982, Boles 1988). In recent years, as escapement in Butte Creek has increased, mortality of oversummering adult fish has also increased due to a combination of high temperatures and the bacterial disease Columnaris (*Flavobacterium columnare*), leading some to suggest that in some years adult carrying capacity has been reached in this stream (Ward et al. 2003).

4.3.4.2 Spawning habitat

The results of habitat suitability studies conducted by the USFWS (2004) indicate that suitable spawning velocities for spring-run Chinook in Butte Creek range between 0.80–3.22 ft/s (24.4–98 cm/s), and suitable substrate size ranges 1–5 in (2.5–12.7 cm) in diameter.

4.3.5 Conceptual model of historical population dynamics

Spring-run Chinook salmon in the Central Valley are generally considered to be stream-type Chinook due to the early arrival of adults and oversummer holding prior to spawning, and the typically long (in some streams at least) juvenile residency period. In this conceptual model, the key feature distinguishing the run is that they migrate in the spring and hold all summer without feeding before spawning in the early fall. By migrating upstream during the high flows of the spring snowmelt, spring-run Chinook salmon historically were able to access higher elevation reaches of the Sacramento River and its tributaries, where cold water allowed them to hold through the summer prior to spawning when temperatures cooled in the fall. We hypothesize that the winter-run Chinook life-history strategy was not possible in some of the tributary basins used by spring-run Chinook (e.g., Deer and Mill creeks) because of summer water temperatures. Spring-run adults hold throughout the summer in deep pools with cover, which helps to keep water temperatures low. In contrast, the summer spawning of winter-run subjects eggs to potentially higher water temperatures in shallow, exposed riffles where ambient air temperatures and solar radiation can increase water temperatures.

There are considerable costs associated with this life history as compared to the fall run, including 3 to 4 months less ocean growth, energy that would otherwise be dedicated to eggs going to fat reserves to allow oversummer holding without feeding, and high predation risk while oversummering. We propose that the prime advantage of this spring-run strategy is being able to reach upstream spawning and rearing habitat that is inaccessible to the fall run, which resulted in spatial segregation of the runs on the spawning grounds, and thus reduced competition for spawning and rearing habitat.

Spawning and rearing habitats that may be accessible to the spring run, but inaccessible to the fall run include (1) areas above falls or obstacles that can not be negotiated during the low flows of summer and fall, and (2) areas above reaches that become too hot for salmon in the summer and fall. During the high spring snowmelt flows, spring-run Chinook can ascend many obstacles that are barriers to upstream migration at lower flows, and can traverse reaches in the spring that will be too warm in the fall for adult salmon.

Under historical conditions, the spring and fall Chinook runs were geographically isolated in terms of where they spawned in the basin, which maintained their genetic integrity. Although spring-run Chinook spawn earlier than fall-run, the timing of spawning of the two runs overlaps enough that hybridization can occur where they share the same spawning areas. Where the spring run is now forced to share spawning grounds in the mainstem Sacramento River with the fall run, fall-run Chinook may dominate because of their longer growth period in the ocean, slightly larger size, and less time spent holding in the stream prior to spawning. Hybridization between the two runs has tended to be to the detriment of the spring-run life history.

In some areas, differences in timing of spawning may still be sufficient to maintain separate runs. Recent improvements to fish passage at Parrott-Phelan Diversion Dam (PPDD) and other downstream dams may have contributed to greater overlap of spring- and fall-run Chinook on spawning grounds in Butte Creek, especially when flows are high in the fall (Hill and Webber 1999, Ward and McReynolds 2001). In recent years, a bar rack has been placed in the PPDD fish ladder to reduce numbers of fall-run Chinook moving upstream of the dam (Ward et al. 2003). Although some fall-run Chinook spawn upstream of PPDD, there appears to be little overlap between the runs in terms of spawning timing, and most fall-run Chinook continue to spawn downstream of the dam (Ward et al. 2003). Peak spring-run spawning in Butte Creek usually occurs during the first week of October, while the peak for the fall run is in mid-to late November (Ward et al. 2003, McReynolds et al. 2005).

The requirement for cool holding pools in the summer limits the spring run to holding in larger mainstem channels, higher elevation streams, or spring-fed streams. The higher-elevation streams generally used by spring-run Chinook for spawning are often characterized by steeper, confined channels with little or no floodplain habitat, although there are some streams used by this run that are not steep and confined. Channels of this type, with high shear stress and sediment transport capacity, are usually coarse-bedded, predominantly cobble and boulder, with gravels typically occurring in small patches where shear stress is locally reduced, such as near large boulders, bank outcrops, or in short, wider reaches. Though spring-run adults may ascend to high elevation reaches to access coldwater pools suitable for summer holding, they may also migrate downstream from these pools to spawn in riffles that they bypassed on their upstream migration.

By holding and spawning in reaches that remain cool all summer, spring-run Chinook have available to them the conditions necessary for extended juvenile residence (i.e., cool summer water temperatures). However, juvenile rearing habitat may be easily saturated by even a small number of successfully spawning salmon. Fry in excess of carrying capacity are likely to disperse downstream, which results in relocation to reaches that are too warm for summer rearing, and requires them to emigrate in the spring and early summer after only a few months of rearing, as age 0+ juveniles, much like juvenile fall-run Chinook. Therefore, rearing habitat limitations may also play a role in regulating spring-run Chinook salmon populations in the Sacramento River basin.

Figure 4.3-2 illustrates the conceptual model.

4.3.6 Effects of anthropogenic changes on spring-run Chinook salmon

Based on the above conceptual model, we would expect that the greatest threat to spring-run Chinook salmon that spawn in the mainstem Sacramento River would be competition with fall-run Chinook. Other sources of mortality, such as poor spawning gravel quality, predation, disease, unscreened diversions, loss of floodplain rearing habitat, and harvest, may also affect the population, but probably to a lesser degree. Whether rearing habitat limitations are also important in regulating spring-run populations is not well understood, and may require further study. Irrespective of questions remaining regarding whether or not certain populations of spring-run Chinook in the Sacramento basin are true to type, most spawning and rearing is currently confined to the tributaries.

4.3.6.1 Effects of Shasta and Keswick dams

Loss of natural spatial segregation between fall and spring runs results in competition for spawning gravels and hybridization

Prior to the construction of dams in the Sacramento and San Joaquin basins, spring-run Chinook salmon migrated during the spring snowmelt flows to access coldwater holding and spawning habitat in the upper and middle reaches of the Sacramento River basin. These steeper, higher-elevation reaches are often characterized by falls and cascades that may be obstacles to upstream movement of salmonids at lower flows. According to our conceptual model, the fall run migrated upstream in the early fall when flows were much lower; therefore, they could not ascend the same obstacles as the spring run to access higher-elevation spawning areas. The fall run typically spawned in the lower reaches of most rivers and streams in the Central Valley (Clark 1929, Hallock and Fry 1967, Reynolds et al. 1993). Thus, the two runs were spatially segregated in the watershed in terms of their spawning grounds even though the timing of their spawning overlapped to some extent (Vogel 1987a, b).

The completion of Keswick and Shasta dams in the mid-1940s blocked spring-run Chinook access to habitat in the McCloud, Pit, and Little Sacramento rivers. After construction of the dams, spring Chinook were forced to spawn in the mainstem Sacramento River below Keswick Dam. Historically, water temperatures would have been too high in the mainstem Sacramento River for spring Chinook salmon to hold in this area during the summer. But because of hypolimnetic releases from Shasta Lake, this reach provides temperatures during the summer that are now suitable for spring Chinook salmon holding and spawning, where before they were only suitable for fall-run spawning once temperatures cooled in the fall. However, coldwater releases from Shasta Dam can warm relatively rapidly during the very hot days typical of the Sacramento Valley in summer and early fall. As a result, both the fall and spring runs must spawn in close enough proximity to Keswick Dam to benefit from these releases. For example, in 2001 over half of fall-run redds (~1,400) were constructed within the first 20 miles (32 km) downstream of Keswick Dam. Spawning habitat in this reach is limited (e.g., 60 gravel patches were used in 2001), and may currently be decreasing as the substrate downstream of the dams coarsens. The elimination of the spatial segregation that had existed between the fall and spring runs results in competition between the runs for the limited spawning habitat. Since fall-run Chinook spawn slightly later than spring-run, spring-run redds may also be superimposed by spawning fall-run fish. This may have contributed to the loss of the spring-run population, along with hybridization between the two runs, as described below.

A lack of spatial segregation between the fall and spring runs following construction of Keswick and Shasta dams contributed to hybridization between the two (Slater 1963, Vogel 1987a, b, Mills and Fisher 1994, Yoshiyama 1998). Similar patterns have been observed in the Feather River, where the spring run historically spawned upstream of the location of Oroville Dam, and where they are now forced to spawn in the same area as the fall run, as well as in the Yuba and American rivers, where forced sympatry on the spawning grounds and subsequent hybridization following dam construction led to CDFG concluding that the spring run was “extinct” in those rivers.

Loss of access to historical holding habitat

The construction of Keswick and Shasta dams also prevented access to the adult holding pools in the McCloud, Pit, and Little Sacramento Rivers that were historically used by the spring run. Spring-run Chinook in the mainstem Sacramento River are therefore forced to hold in pools that are more accessible to anglers than those in the high-gradient reaches that adults used historically. As a result, angling pressure on spring-run adults may have increased as an indirect effect of the displacement caused by dam construction. These factors may contribute to greater mortality of pre-spawning adults, and possibly affect viability of eggs in females exposed to high temperatures.

4.3.6.2 Effects of Red Bluff Diversion Dam

At the Red Bluff Diversion Dam (RM 243), gates that allow for upstream fish passage are currently closed in early May, potentially blocking spring-run Chinook salmon access to spawning habitat upstream. Historically, spring-run Chinook migrated upstream in the Sacramento River through May and into June (Rutter 1908). Artificial selection against later-returning fish could reduce genetic diversity and reduce the species’ resilience and adaptability to future changes in climate and hydrologic regimes, whether natural or anthropogenically induced.

4.3.6.3 Effects of hatchery practices

Hybridization between spring- and fall-run Chinook salmon runs

Hatchery practices in the 1960s included the spawning of spring and fall runs together, resulting in the release of hundreds of thousands of hybrids. At the Feather River Hatchery, operators attempted to avoid hybridization of the spring run by assuming that all salmon taken at the hatchery in September were spring-run, and that all fish taken after 15 October were fall-run (Cramer and Demko 1997). However, no Chinook were collected prior to September due to concerns that earlier caught fish might die at the hatchery prior to spawning. Thus, by the time fish were collected, both spring and fall stocks were present, and indistinguishable. Fall and spring run stocks were inadvertently hybridized and now form one hatchery strain (Cramer and Demko 1997). Hybrids from the Feather River Hatchery have been widely released, and have also subsequently strayed throughout the basin, including in the mainstem Sacramento River upstream of its confluence with the Feather River (Cramer and Demko 1997). Currently, only the Feather River Hatchery propagates “spring-run” Chinook, but this hatchery stock is assumed to be completely hybridized with the fall run.

Despite evidence of hybridization between the fall and spring runs below Keswick Dam and in the Feather River Hatchery, genetic analysis has to date not conclusively supported the hypothesis that the Central Valley spring run has been largely lost to hybridization with the fall run (Hedgecock et al. 2001). One challenge to resolving this question is that no historical allozyme data are available for naturally spawning Sacramento River spring-run Chinook salmon against which current allele frequencies may be compared (NMFS 1999a). Banks et al. (2000) conducted an analysis of microsatellite DNA from Central Valley Chinook salmon to evaluate genetic diversity within and among the four runs. Butte, Deer, and Mill creek spring-run samples were analyzed in this study. Previously, it was generally believed that the spring-run Chinook spawning in Deer and Mill creeks represented the stocks closest in genetic makeup to historical populations in the basin (Cramer and Demko 1997). In addition, it was believed that the subpopulation in Butte Creek had undergone substantial hybridization with fall-run Chinook stock from the Feather River Hatchery (Yoshiyama et al. 1998). Somewhat surprisingly, Banks et al. (2000) found no evidence that the runs in these tributaries had hybridized with fall-run Chinook—the proportions of microsatellite genotypes found conformed to those expected from random mating among individuals. Banks et al. (2000) also found evidence of two distinct lineages of spring-run Chinook: the Butte Creek subpopulation clusters farther from the fall run than subpopulations in Deer and Mill creeks, whereas it would have been expected to be closer under previous assumptions regarding hybridization.

Whether any true spring-run Chinook salmon remain in the mainstem Sacramento River is debatable, with many experts concluding that competition and hybridization with the fall run have eliminated the last of the Central Valley spring run in the mainstem Sacramento River (Moyle 2002).

Recent genetic analysis suggests that (1) wild spring-run Chinook populations from different streams in the Sacramento River are more closely related than to each other than to fall-run Chinook within the same basin, and (2) naturally spawning and hatchery fish of the Feather River are more closely related to each other and to the Feather River fall run than they are to the three wild-type spring-run populations (NMFS 2003, Hedgecock 2002, as cited in NMFS 2003).

Change in timing of upstream migration

Increased hybridization between fall and spring Chinook salmon has reduced temporal segregation between the runs in some areas, thus increasing overlap on the spawning grounds and subsequent hybridization (CDFG 1990). In the Feather River, the time of river entry for "spring-run" Chinook salmon has apparently shifted to later in the season, and is now intermediate between timing of entry of spring run into other tributaries and timing of entry of the fall run. Whereas wild-type spring-run Chinook enter Deer and Mill creeks primarily in mid-April to mid-June, coded-wire tag data and anecdotal information from anglers indicates that Feather River fish do not enter fresh water until June or July (Cramer and Demko 1997).

Identification of the spring run at Red Bluff Diversion Dam is complicated by their small numbers. Also, they cannot always be distinguished by visual characteristics or by run timing (due to the naturally protracted return time of the now-more-abundant fall run). However, analysis of data from the 1970s and 1980s by Cramer and Demko (1997) indicated that, since 1970, the passage of "spring-run" Chinook past Red Bluff Diversion Dam has gradually shifted later by over a month, moving the timing of spawning closer to that of the fall run. By the 1990s, run timing was indistinguishable between the spring run and fall run at Red Bluff Diversion Dam, likely because of genetic introgression (hybridization) between the two. By 1995, only a few redds were observed in the mainstem Sacramento River during the time when spring-run spawning was initiated in tributaries (August and September).

4.3.6.4 Effects of ocean harvest

Ocean harvest may have altered both the age at which spring run now return to spawn and the fecundity of fish that reach spawning grounds. Spring-run Chinook salmon used to return to spawn predominantly at ages 4 and 5 at a large size, but now return primarily at ages 3 and 4. Ocean harvest removes the largest fish from the population and selects for fish that spend fewer years at sea, because the longer a subadult salmon remains in the ocean to feed and grow, the more likely it is to be harvested. Minimum size limits also select against larger, older salmon. Artificial selection for fish that return at younger ages and smaller sizes may affect the population in several ways. Fecundity is usually directly related to the size of adult females, so smaller fish will lay fewer eggs. Larger females can spawn in larger-sized spawning gravels, so smaller adults may be restricted to spawning in smaller gravels. Smaller females may also dig shallower redds that are more vulnerable to scour or to superimposition in areas where fall-run Chinook spawn. Reducing the number of years that a cohort returns to spawn also increases the vulnerability of the population to stochastic events. In 1994, it was reported that 87% of spring Chinook returned at age 3 (Fisher 1994); however, more recent estimates from Butte Creek based on coded-wire-tag data indicate that a large portion are returning at age 4 (approximately 69% in 2003 (Ward et al. 2004b), perhaps because of restrictions on harvest following their listing as a threatened species.

4.3.7 Management implications, key hypotheses, and uncertainties

Based on conditions in the mainstem Sacramento River with respect to the ongoing lack of spatial and temporal segregation on the spawning grounds, restoring the spring-run population to the mainstem Sacramento River is not likely to be successful, and may cause challenges in the management of the fall-run population. Management actions for spring-run Chinook salmon that are focused on preserving the wild-type runs in key tributaries such as Butte, Mill, and Deer creeks are more likely to be successful at preserving the integrity of this run in the Sacramento basin.

4.3.7.1 Restore spatial segregation between the runs by managing fish passage at artificial barriers

If resource managers choose to pursue the restoration of a mainstem-spawning population of spring-run Chinook, then the lack of spatial segregation between the spring and fall runs in the mainstem Sacramento River is a primary challenge to spring-run conservation efforts. To establish a spring run in the main stem, management efforts would need to focus on establishing spatial segregation between the runs, either through use of an existing, or new fish barrier that will allow for selective passage of spring-run adults to upstream areas. The ACID Dam (RM 298.4) could be used as such a barrier. Spring Chinook could be allowed to pass upstream to spawn above the dam, and the ladder could be closed later in the season to force fall-run Chinook to spawn downstream of the dam. This action could decrease competition and hybridization between the two runs; however, not allowing passage to fall-run Chinook above the ACID Dam could increase redd superimposition below the dam. In addition, spring-run production would be limited by the amount of habitat available upstream of ACID Dam, which can change over time as high flows scour gravel and route it downstream of the dam. Periodic gravel augmentation would be required for the reach above ACID Dam to maintain or expand spawning habitat for a restored spring-run salmon population.

4.3.7.2 Modify gate operations at Red Bluff Diversion Dam

Keeping the gates at the Red Bluff Diversion Dam open until late May would allow more spring-run Chinook salmon access to areas upstream through inclusion of later-returning fish. This measure will become more important as efforts increase to establish a significant population of spring-run salmon in Battle Creek.

4.3.7.3 Create a controlled flood bypass managed specifically to promote fry rearing

As stream-type salmon, a fraction of spring-run juveniles may spend a summer rearing in natal streams before emigrating to the ocean. After emergence, spring-run juveniles display agonistic behavior, establishing and defending territories. This behavior means that summer rearing habitat can be quickly saturated, even if escapements are low, because of the area required to support each juvenile. Spring-run that migrate downstream as fry often represent those individuals displaced as a result of rearing habitat saturation in upstream reaches. Because these fry are forced to migrate downstream at a small size < 1.6 m (40 mm), they are vulnerable to predation, such that the fry component may not contribute significantly to future escapements. However, recent research conducted on the Butte Creek population of spring-run salmon suggest that successful rearing by spring-run fry in the Sutter Bypass may be stimulating the recent increase in escapements. Generally, the Deer and Mill creek populations of spring-run do not seem to have the same success in fry rearing. To improve fry rearing potential for the Deer and Mill Creek populations, we recommend the creation of a dedicated floodplain/bypass area along the mainstem Sacramento River downstream of Deer and Mill creeks. A bypass in the vicinity of Deer and Mill creeks would provide rearing habitat to fry and juveniles outmigrating to the main stem from these important spawning tributaries for remaining wild-type spring-run Chinook. Such a bypass should be constructed to provide high-quality rearing habitat at relatively low flows, so that the habitat is available for a large portion of every winter, even during drier years.

4.4 Fall-Run Chinook Salmon

The fall run of Chinook salmon is the most abundant and widely distributed in the Central Valley, in large measure because it has suffered relatively less displacement from historical habitats by dam construction. The relatively high abundance of fall-run Chinook salmon (or "fall Chinook") is also a function of hatchery supplementation, because fall Chinook have been the primary target of hatchery production at Central Valley hatcheries for several decades. Despite the significantly higher abundance of fall Chinook relative to other salmonid populations, escapements have generally declined over the past few decades, and NMFS designated the Central Valley Fall (and Late-fall) Chinook salmon ESUs as a species of concern in 2004 (NMFS 2004c). As the most abundant salmonid species in the Central Valley, fall Chinook constitute an important component of the commercial and recreational salmon fishery in California.

4.4.1 Geographic distribution

Within the range of the Central Valley ESU, large populations of fall-run Chinook salmon are found in the Sacramento River and its major tributaries. Fall Chinook are the most widely distributed salmonid in the Sacramento River basin, with significant spawning populations documented as far north as the upstream limit of anadromy in the upper Sacramento River (e.g., Keswick Dam at RM 302) and as far south as the American River near Sacramento. Sizeable spawning populations occur in other tributaries to the Sacramento River (e.g., Clear Creek, Battle Creek, Butte Creek, Feather River), with more modest spawning populations on numerous smaller tributaries (e.g., Deer, Mill, Cow, and Antelope creeks). Self-sustaining runs are also found in tributaries to the San Joaquin River, including the Merced, Tuolumne, Stanislaus, and Mokelumne rivers.

Currently, the upstream limit of spawning is generally dictated by the presence of dams (e.g., Keswick Dam on the upper Sacramento River), weirs (e.g., the fish barrier at Coleman National Fish Hatchery on Battle Creek) or flow-related passage barriers located in the tributaries (e.g., Clear Creek canyon reach). Until 2001, the ACID Dam (RM 298.4) generally impeded the passage of fall Chinook, thereby forcing spawning to occur downstream; however, recent improvements to the dam's fish passage facilities have opened the reach between ACID Dam and Keswick Dam (RM 302) to fall Chinook spawning in the mainstem. In the smaller tributaries, the upstream limit of fall Chinook spawning can vary each year, because variable hydrologic and climatic conditions, and water diversions can affect the location of flow-related passage barriers or suitable water temperatures that support fall Chinook spawning.

Fall Chinook spawning has been documented throughout the gravel-bedded reach of the mainstem Sacramento River down to Colusa (RM 143); however, few redds are recorded downstream of Princeton (RM 163), which is the downstream limit of annual redd surveys conducted by CDFG. These annual redd surveys also indicate that the bulk of fall Chinook spawning occurs upstream of Cottonwood Bridge (RM 273) (Snider et al. 2000a).

The full length of the mainstem Sacramento River between Keswick Dam (RM 302) and the Delta provides a migration corridor for adult upstream migrants and juvenile emigrants. As fall Chinook fry and parr migrate downstream, they also use the lower reaches of non-natal tributaries as rearing habitat (Maslin et al. 1997). During periods of high winter and spring runoff, fall Chinook juveniles are also diverted into the bypasses that border the Sacramento River, where growing conditions are generally better than mainstem rearing habitats, which can facilitate higher rates of juvenile survival (Sommer et al. 2001a). Natural floodplain or riparian areas that become inundated during high flows may also provide good habitat for juvenile Chinook salmon and prevent them from being displaced downstream (Limm and Marchetti 2003).

4.4.2 Population trends

Spawning populations of fall Chinook in the Sacramento River basin belong to the Central Valley Fall- and Late-Fall Evolutionary Significant Unit (ESU) as delineated by NMFS. It is important to note that NMFS combines fall-run Chinook salmon and late-fall-run Chinook salmon within a single ESU. This chapter addresses fall-run Chinook salmon, and Chapter 4.5 addresses late-fall-run Chinook salmon separately.

Though NMFS considers fall-run and late-fall-run Chinook salmon as part of the same ESU in the Central Valley, most resource agencies track the two runs separately. For example, CDFG conducts aerial redd surveys that specifically target late-fall-run salmon, and the AFRP tracks late-fall-run salmon escapements as a separate population in its population monitoring database. However, reports on fall-run escapement estimates vary, because some include late-fall-run in the estimates, while others do not. Because the reports often fail to clarify which runs are being enumerated in the escapement estimate, care must be exercised when using fall-run escapement estimates, especially from different sources.

Sacramento River Basin. For fall Chinook salmon populations in the Sacramento River basin (including the Feather and American rivers), the Pacific Fishery Management Council (PFMC) has defined a conservation target of 122,000 to 188,000 spawners each year, which includes both naturally spawning adults (of both natural and hatchery origin) and those harvested for hatchery operations. Because a CDFG study indicates that as much as 25% of fall Chinook adults returning to the Sacramento River basin are harvested as part of the sport fishery (PFMC 2006), escapements must generally range between 152,000 and 235,000 adults to satisfy the PFMC conservation target for spawners. This conservation target has been achieved since 1994, and fall Chinook stocks in the Sacramento River basin have been rebounding since the low escapement year of 1992, when approximately 81,000 adults returned to spawn. Annual escapements since 1992 have averaged 369,000 adults, but during that time, there have been some banner years, including 2001 (546,056), 2002 (775,499), and 2003 (521,625) (Figure 4.4-1). Because most fall Chinook return to spawn as 3-year-olds, the high escapement year of 2002 (775,499) resulted in state and federal biologists predicting an escapement of 983,600 for 2005; however, only 383,500 fall Chinook adults returned to the Sacramento River basin to spawn that year (PFMC 2006). A spike in the number of grilse (2-year-olds) that returned to spawn in 2004 contributed to the expectation of higher escapements in 2005 (PFMC 2006). Agency biologists also anticipated that ocean harvest restrictions designed to protect Klamath River salmon stocks would benefit salmon stocks from the Sacramento River basin by reducing adult mortality, thereby contributing to a higher escapement in 2005. It is not yet clear why fewer adults returned in 2005 than were predicted, but poor ocean rearing conditions associated with the Pacific Decadal Oscillation (PDO) may have contributed to the lower-than-expected escapement in 2005 (Varanasi 2005).

The hatchery component of fall Chinook escapements in the Sacramento River basin can be significant, especially in the Sacramento, Feather, and American rivers where hatcheries are located. On average, more than 25,000 of the adults that return to spawn each year are of hatchery origin (Cramer and Demko 1997).

Upper Sacramento River. The number of fall Chinook adults that return to the upper Sacramento River system (i.e., the mainstem channel and tributaries located upstream of the Feather River confluence) is similar to the pattern for the whole Sacramento River basin. Upper Sacramento River escapements exhibited a similar low in 1992 (37,990), but the years surrounding 1992 (1989–1994) also had relatively low escapements (Figure 4.4-2). It is interesting to note that this period of low escapement generally coincides with a multi-year drought during the late 1980s and early 1990s. Beginning in 1995, fall Chinook escapements began to improve, totaling more than 100,000 returning spawners each year, and eclipsing the 200,000 mark in nearly half of those years (Figure 4.4-2). Figure 4.4-2 also illustrates the increasing harvest of adult fall-run salmon for hatchery operations, reflecting an increase in hatchery production in recent years.

Mainstem Sacramento River. Escapements that are aggregated over the entire Sacramento River basin can mask changes in the population of fall Chinook that spawns in the mainstem river, primarily because escapements to tributaries where hatcheries are located (e.g., Battle Creek, Feather River, American River) can have large numbers of adults returning in years when numbers of natural spawners in the mainstem are low. For example, during the high escapement year of 2002, about 71,687 fall Chinook spawned in the mainstem river between Princeton (RM 163) and Keswick Dam (RM 302), but more than 463,000 adults returned to Battle Creek to spawn or to be harvested at Coleman National Fish Hatchery (CNFH) (Figure 4.4-3). Similarly, in 1998, only 6,318 adults spawned in the mainstem channel, even though more than 98,000 fall Chinook returned to Battle Creek (Figure 4.4-3). The dramatic increase in the number of fall-run salmon returning to Battle Creek reflects the influence of hatchery operations at CNFH and the degree to which hatchery production influences overall escapements.

Escapements for Chinook salmon populations in the Central Valley are naturally variable, reflecting changes in inter-annual environmental conditions (e.g., hydrology, ocean conditions) and shifts in the sources and rates of mortality caused by human activities (e.g., entrainment in Delta pumps, changes in ocean harvest restrictions). Escapements for the population of fall Chinook that spawns in the mainstem Sacramento River have been especially variable since 1997, ranging from 5,718 to 133,365 returning adults (Figure 4.4-3). Part of this variability can be explained by the low escapement year of 1998 (5,718) and the propensity for fall Chinook to return as 3-year-olds. With so few adults spawning in 1998, the returning class of 2001 experienced a dip in escapements (57,792), which also affected then number of adults returning in 2004 (34,050) (Figure 4.4-3). Though the successive classes of the 1998 parent class rebounded from the low of 5,718 adults, the pattern of escapements shows the lasting effects that a single year-class crash can have. It is not clear what factors precipitated the crash of the 1998 class of spawners.

4.4.3 Life history

Adult fall-run Chinook salmon migrate into the Sacramento River and its tributaries from June through December in mature condition, with upstream migration peaking in September and October. Fall Chinook adults spawn soon after arriving at their spawning grounds between late September and December, with peak spawning activity in late October and early November (Yoshiyama et al. 1998, Table 4.4-1). Fry emergence occurs from December through March, and fry rear in freshwater for only a few months before migrating downstream to the ocean as smolts between March and July (Yoshiyama et al. 1998); consequently, fall Chinook are “ocean-type” salmon, because juveniles usually migrate to sea during their first year of life, and because adults do not spend much time in freshwater before spawning (Healey 1991, Moyle et al. 1989). Fall Chinook spend most of their life in coastal ocean waters before returning to their natal river to spawn, most often as 3-year-olds.

Juvenile Chinook salmon feed and grow as they move downstream in spring and summer; larger individuals are more likely to move downstream earlier than smaller juveniles (Nicholas and Hankin 1989, Beckman et al. 1998), and it appears that in some systems juveniles that do not reach a critical size threshold will not outmigrate (Bradford et al. 2001). Bell (1958, as cited in Healey 1991) suggests that the timing of yearling smolt outmigration corresponds to increasing spring discharges and temperatures. Kjelson et al. (1981) observed that peak seine catches of Chinook fry in the Sacramento-San Joaquin Delta correlated with increases in flow associated with storm runoff. Flow accounted for approximately 30% of the variability in the fry catch. Photoperiod may also be important, although the relative importance of various outmigration cues remains unclear (Bjornn 1971, Healey 1991).

When fall-run Chinook salmon produced from the Sacramento-San Joaquin system enter the ocean they appear to head north, and rear off the northern California-southern Oregon coast (Cramer 1987, as cited in Maragni 2001). Fall-run Chinook salmon typically have a greater tendency to remain along the continental shelf than do stream-type Chinook (Healey 1983, as cited in Quinn 2005). Ocean conditions are likely an important cause of density-independent mortality and inter-annual fluctuations in escapement sizes.

Table 4.4-1. Life history timing of fall-run Chinook salmon in the California Central Valley (from Vogel and Marine 1991).

Life stage	**Month**																							
	Jan		**Feb**		**Mar**		**Apr**		**May**		**Jun**		**Jul**		**Aug**		**Sept**		**Oct**		**Nov**		**Dec**	
Adult migration past Red Bluff Diversion Dam													░	░	░	░	▒	▓	▓	▓	▒	░	░	░
Spawning																		░	▒	▓	▓	▒	░	░
Incubation	▒	▒	▒	▒	░	░													░	░	▒	▒	▒	▒
Fry emergence	▓	▓	▓	▓	▒	▒																	▒	▒
Rearing in mainstem[1]	▒	▒	▒	▒	▒	▒	▒	▒	▒	▒	▒	▒												
Outmigration past Red Bluff Diversion Dam	░	▒	▒	▒	▒	▓	▓	▓	▓	▒	▒	░												
Entry into Sacramento-San Joaquin Delta	▒	▒	▒	▒	▒	▒	▒	▒	▒	▒	▒	▒												

[1] A few fall-run Chinook salmon may remain upstream of Red Bluff Diversion Dam to rear to a yearling life stage.

░	Period of light activity
▒	Period of moderate activity
▓	Period of peak activity

4.4.4 Habitat requirements

General habitat requirements for Chinook salmon are described in Section 4.1. Only habitat requirements specific to fall-run Chinook salmon are described here.

4.4.4.1 Spawning habitat

Chinook salmon are capable of spawning within a wide range of water depths and velocities, provided that intragravel flow is adequate (Healey 1991). Fall Chinook salmon are generally able to spawn in deeper water with higher velocities because of their larger size (Healey 1991).

4.4.4.2 Fry rearing habitat

In the mainstem Sacramento River, rearing habitat for fall Chinook fry would be available in eddy zones downstream of point bars and in velocity shear zones where the thalweg crosses from one bank to the opposite bank. The eddies and velocity shear zones provide juveniles with slow-velocity water to reduce the energy required for a juvenile to maintain position and adjacent high-velocity water to deliver aquatic insect drift.

Research indicates that shallow water habitats can promote faster growth of juvenile Chinook than deep water areas (Sommer et al. 2001a), most likely because of warmer water temperatures and higher prey densities (Limm and Marchetti 2003, Stillwater Sciences 2003). In the Sacramento River system, seasonally inundated shallow water habitats can be found within the bankfull channel during periods of elevated discharge in the winter and spring, when features associated with point bar complexes are inundated. During years with high winter and spring flows, shallow water rearing habitat can also be created through floodplain inundation and inundation of the many flood bypasses that border the mainstem Sacramento River. Faster growth of juvenile salmon can increase survival by increasing the range of prey available to them, by reducing their vulnerability to predation by gape-limited predators such as piscivorous fish (Myrick and Cech 2000), and by improving their ability to compete with other salmonids for food and space. Sommer et al. (2001a) found that juvenile Chinook released into the Yolo Bypass had higher adult return rates than those released in the mainstem. Limm and Marchetti (2003) note that changes to the Sacramento River's natural hydrograph have decreased connectivity between off-channel habitats and the mainstem in the spring. They observed stranding of juvenile salmon in off-channel ponds as the ponds became disconnected from the main channel. Off-channel ponds may also provide suitable year-round habitat for non-native species such as largemouth bass that prey on juvenile salmon. Stranding and predation mortality could outweigh the benefits associated with juvenile salmonid use of off-channel rearing habitats. The magnitude at which stranding of juvenile salmon occurs in the flood bypasses and natural off-channel habitats has not been adequately assessed to date. Stranding studies are inherently difficult, as stranded fry and juvenile salmon are difficult to survey, especially in vegetated habitats, and many bird, mammal, and fish predators may eat them or carry them off if they become stressed by high temperatures or after stranding. Stranding potential in the Yolo Bypass may be reduced because it is graded for agriculture and stage decreases are relatively gradual (Sommer et al. 2001a). Although some relatively minor avian predation may occur in shallow habitats like the bypasses, substantial predation by piscivorous fish is unlikely because of the shallow and ephemeral nature of these habitats.

4.4.5 Conceptual model of historical population dynamics

The life history features that distinguish fall-run Chinook salmon from other anadromous salmonids in the Sacramento River are timing of entry from the ocean (late summer/fall), spawning habitat distribution (lower on the river than other runs), timing of spawning (fall) and timing of fry emergence (late-winter). Figure 4.4-4 illustrates the key components of the fall Chinook life history strategy.

The mass outmigration of fall Chinook smolts is believed to act as a predator-swamping strategy. Although many other salmonids and races of Chinook salmon smolt at larger sizes than fall-run Chinook salmon, few outmigrate in the magnitude of the fall run. Fall Chinook generally spawn in the lower alluvial reaches of mainstem rivers where spawning habitat is more abundant than in the higher-elevation, steeper gradient tributaries where winter-run and spring-run Chinook spawn. They also spawn at a time when air and water temperatures are decreasing (late fall), which grants them access to spawning gravels located farther downstream. Fall Chinook adults can also spawn in the coarse gravels that compose the bed of mainstem rivers because of their relatively large size, which can reach weights up to 99 lbs (45 kg). By utilizing abundant spawning habitat to produce large numbers of offspring, fall run Chinook may effectively swamp their predators during juvenile outmigration.

Because they spawn in lower mainstem reaches of large rivers, where water temperatures may increase rapidly in the spring and summer, fall Chinook fry must emigrate quickly from fresh water at a relatively small size < 3.5 in (90 mm) before water temperatures become stressful or lethal.

There are a variety of early-fry rearing strategies, but fall Chinook generally exhibit two rearing strategies: migrating to the lower river or estuary as fry, or remaining to rear in the gravel-bedded reach for about three months and then smolting and outmigrating. The highest abundances of fry in the Delta are observed in wet years (Brandes and McLain 2001). Both of these rearing strategies are made possible by the timing of fry emergence between January and March and by the location of spawning in mainstem rivers. The fry that rear in the gravel-bedded reach of the mainstem enjoy better food supplies and relatively less competition with juveniles of other Chinook runs. Most winter- and spring-run juvenile Chinook will have already outmigrated from the mainstem by March (Vogel and Marine 1991), and late-fall-run juveniles will be farther upstream or will be smaller because they emerge later than the fall-run. Similarly, rearing habitat is generally more abundant in mainstem rivers, as compared with higher-elevation tributaries where winter-run and spring-run juveniles begin rearing, so fry rearing habitat is not likely to be limiting. Fall Chinook fry also rear during a time and in the location where floodplain inundation is most likely to occur, thereby expanding the amount of rearing habitat available. A greater supply of food and reduced competition in mainstem rearing habitats allows fall Chinook to have relatively higher growth rates than the juveniles of other Chinook runs. As a result, fall Chinook juveniles can reach smolting size (~3.5 in [90 mm]) in a relatively short period of time and emigrate from freshwater in the spring before water temperatures become lethal. By growing relatively quickly to smolt size, fall Chinook leave fresh water before they reach a size when both their food demands and competition for summer rearing habitat increase, both of which may increases the potential for density-dependent mortality.

Early emergence and fry migration to the lower mainstem, the flood bypasses, or the estuary, is a viable life-history strategy because they can rear in these downstream habitats while water temperatures are still suitable and while there is relatively less competition from the juveniles of other runs. Rapid growth in these downstream reaches and the estuary allow fall Chinook fry to reach smolting size before summer water temperatures become too high. However, relative survival of fry appears to be higher in the upper Sacramento River than in the Delta or bay, especially in wet years (Brandes and McClain 2001).

One potential disadvantage of early emergence and emigration and rearing in mainstem channels and the estuary is the possibility of higher predation mortality because of the relatively small size of emigrants. However, fall Chinook fry exhibit several characteristics to combat predation mortality. Predators often occupy deep pools in mainstem channels, so fry generally use shallow water habitat found along channel margins or in runs and riffles to avoid predators. Because rearing habitat is not limiting for fall Chinook fry, they do not exhibit territorial behavior, which allows them to rear, smolt, and outmigrate in higher densities. By emigrating synchronously in schools, rather than as individuals, fall Chinook fry and smolts can swamp potential predators to avoid significant losses to predation, and by emigrating in late spring, they have the advantage of higher discharge fueled by early snowmelt, which can reduce their exposure to predation.

By producing large numbers of smolts, fall Chinook enjoy relatively high escapements in a positive feedback loop. As a result, the competition for spawning habitat is the most likely source of density-dependent mortality for fall Chinook, primarily as a function of redd superimposition whereby later arriving females dig redds on top of existing redds, causing substantial mortality of the previously-deposited eggs (McNeil 1964b, Hayes 1987). In general, redd superimposition confers an advantage to late spawners because their progeny will suffer less egg mortality as the number of subsequent spawners dwindles. However, late spawners can pay a penalty of poorer smolt survival in the subsequent spring if their progeny emerge too late to emigrate from freshwater before water temperatures get too high. Because they emerge later, the progeny of late spawners will also have less time to grow before they need to emigrate, so that their relatively smaller size exposes them to higher rates of predation than fry that emerged earlier in the winter. Thus, the timing of fall Chinook spawning and emergence is constrained at the beginning of the season by low flows and warm temperatures and by redd superimposition by later-arriving spawners, and at the end because of warming water temperatures in the spring that would cause mortality of smolts.

The success of the fall Chinook life history strategy is predicated on the production and survival of high numbers of juveniles, which requires abundant spawning habitat. As a result, the production of fall run Chinook salmon is likely limited by available spawning habitat. As discussed below, anthropogenic alterations to the amount of available spawning habitat pose the greatest risk to fall Chinook salmon.

4.4.6 Effects of anthropogenic changes on fall-run Chinook salmon habitat

Based on the above conceptual model, it is expected that the greatest potential anthropogenic threats to fall-run Chinook salmon would be reduction in spawning habitat or increased density independent mortality at subsequent life-stages. Other factors, such as spawning gravel quality (e.g., particle size distribution, fine sediment deposition), increased risk of predation, unscreened diversions, hybridization with spring-run Chinook salmon, and ocean harvest may also affect population dynamics, but likely to a lesser degree. Each of the primary threats to the population is discussed below.

4.4.6.1 Spawning habitat

Anderson-Cottonwood Irrigation District (ACID) Dam. The construction of the ACID Dam (RM 298.4) near Redding in 1916 likely caused delays in the upstream migration of fall Chinook salmon. ACID Dam was operated seasonally, typically between April and October, so the flashboards were often in place during the beginning and peak of fall Chinook migration upstream migration in late September. Though the delays may not have caused direct mortality, they may have contributed indirectly to adult mortality by exposing spawners to increased angling pressure as they congregated downstream of the dam. Because fall Chinook typically spawn soon after entering freshwater, any delay in reaching upstream spawning areas until later in the fall may have reduced spawning success. The dam may have also forced some upstream migrants to spawn downstream of the dam, which may have increased redd superimposition. Following construction of the dam, observers noted lower escapements of Chinook salmon in the upper reaches of the Sacramento River and its tributaries (Yoshiyama et al. 2001).

Keswick and Shasta dams. Because they typically spawn in the lower-elevation reaches of large mainstem rivers, fall Chinook have suffered the least displacement from historical habitats as a function of dam construction. Nevertheless, fall-run Chinook salmon were documented in reaches up to 1,000 ft (300 m) elevation on the McCloud River prior to the construction of Keswick and Shasta dams (H. Rectenwald and R. Yoshiyama pers. comm., as cited in NMFS 1999b). Accounts of available spawning habitat upstream of the Shasta Dam site (Hanson 1940) also suggest that fall-run Chinook salmon may have experienced the greatest absolute loss of spawning area of all the Chinook salmon runs in the Sacramento River basin, even though the percentage of spawning habitat lost was low relative to the other runs. However, overall the fall-run has suffered less than other runs, in part because they spawn during fall when air and water temperatures are declining, and thus they can use spawning habitats farther downstream than other runs. Though much of their spawning occurs where other runs spawn (e.g., RM 273 to RM 302), they also spawn down to Princeton (RM 163). Although the abundance of the fall Chinook has declined, it has been far less dramatic than the escapements of other runs, in part because they have more available spawning habitat.

Bed coarsening has likely reduced fall Chinook spawning habitat, as high flow releases from Shasta Dam recruited gravel stored in the channel bed, leaving behind larger lag particles that cover an increasing percentage of the channel bed surface. Again, the effects of bed coarsening on fall-run have likely been less than the effects on the other runs of Chinook salmon that spawn in the mainstem because fall Chinook can utilize gravel resources located farther downstream in the mainstem channel.

Red Bluff Diversion Dam. Beginning in 1967, Red Bluff Diversion Dam (RBDD) may have impeded or prevented access to upstream spawning habitats. Because fall-run Chinook salmon spawn in the fall when air and water temperatures decline, spawning was likely still possible below the dam. However, spawning habitat below Red Bluff Diversion Dam is more susceptible to increased fine sediment concentrations because of sediment supplied from tributaries. In addition, the bed load below Shasta Dam has been coarsening over time as a result of decreased gravel supply, further restricting available spawning habitat. The progeny of adults that spawned downstream of Red Bluff Diversion Dam are also more susceptible to the potential for elevated water temps during low-flow conditions. Gate operations were changed at RBDD beginning in the winter of 1986 (Table 4.2-1) so that gates are usually raised beginning in mid-September, thereby reducing the impact on fall Chinook upstream migration.

4.4.6.2 Fry and juvenile rearing habitat

Under current conditions, the mortality of fry and smolts is higher than would have occurred historically. Because emergent fry usually rear in shallow-water areas associated with channel margins, and because they migrate downstream as spring irrigation demands increase, they are susceptible to entrainment in water diversions. An increase in the abundance and distribution of exotic predators (e.g., largemouth bass) in the lower Sacramento River has also likely increased predation mortality for fall Chinook smolts and muted the benefits of the “swamping” strategy. In addition, prior to the channelization of the Sacramento River, rearing habitat for juvenile

Chinook salmon would have been much more abundant due to the availability of abundant floodplain, side-channel, and off-channel habitats.

4.4.6.3 Hybridization

Historically, spring Chinook and fall Chinook both spawned during the fall, but they were separated spatially because spring Chinook salmon spawned in upper tributaries that the fall run could not access. Under current conditions the Keswick and Shasta dams have prevented spring Chinook salmon from accessing upper tributaries, and instead they spawn in the mainstem Sacramento River where the fall run spawns. The elimination of spatial segregation of fall Chinook and spring Chinook spawning contributed to hybridization occurring from co-mingling during spawning (Yoshiyama et al. 1998). Also, hatchery practices have likely mixed fall and spring Chinook stocks, causing even greater hybridization. By hybridizing with spring Chinook, the peak spawning activity of fall Chinook has likely shifted to occur earlier than it did historically.

4.4.6.4 Hatcheries

Fall-run Chinook salmon have long been a focus of hatchery production in the Central Valley, and the artificial propagation of fall-run salmon supports the commercial and recreational harvest of salmon in California. Within the Sacramento River basin, CNFH produces substantial numbers of fall-run salmon for release in the Sacramento River and Bay-Delta estuary. Increased escapements of fall-run salmon to Battle Creek in recent years (Figure 4.4-3) suggest that hatchery operations are having a strong influence on the population. The release of hatchery Chinook salmon may have negative effects on rearing fall-run Chinook due to increased competition for food and space; however, little is known regarding the effects of hatchery releases on wild juvenile Chinook in the Sacramento River.

Strays from Feather River may constitute 40% of the fall-run salmon that migrate upstream of RBDD to spawn in the mainstem Sacramento River and tributaries (Cramer and Demko 1997).

4.4.7 Management implications, key hypotheses, and uncertainties

4.4.7.1 Assess redd superimposition in upstream reaches

The degree of redd superimposition mortality that occurs currently for the fall-run Chinook salmon population is unknown. Due to fall run Chinook salmon spawning further downstream than other runs, the potential for intra-specific superimposition is less than for other runs. However, spatial segregation is decreased with barriers to distribution, and spawning habitat is likely the only density dependent source of mortality for fall run, and with large escapement habitat is potentially limiting production. In addition, the progressive coarsening of bedload downstream of Shasta Dam is potentially decreasing available spawning habitat. A redd superimposition study is recommended to address these uncertainties, and exploring gravel augmentations may be warranted in the future.

4.4.7.2 Increase spawning habitat

If the redd superimposition study suggests significant egg mortality due to redd superimposition, then one approach for increasing spawning habitat is to increase the frequency of gravel augmentation in the upper Sacramento River. As discussed in Chapter 4.2, more than 242,000

yd^3 (185,022 m^3) of spawning-size gravel have been added to the mainstem channel to date, and these gravel injections have likely played an important role in maintaining local patches of existing spawning habitat. However, the periods between gravel augmentation efforts (10 years) have been long. To compensate for the cumulative loss of gravel since the completion of Shasta Dam, a gravel augmentation program should emphasize injecting larger volumes of gravel to the channel than have been added to date.

Another potential technique for expanding spawning habitat would be to remove the coarse surface layer from armored reaches. Though the channel bed of the upper Sacramento River has been coarsening downstream of Keswick Dam (RM 302) since the completion of Shasta Dam, the armor layer on the channel bed surface traps finer sediments stored in the subsurface. Removal of the coarse surface layer can expose these finer sediments to spawning salmon. Exposing the finer sediment in the subsurface can also expose it to scour and transport during high flow events, so removal of a coarse surface layer should be implemented in conjunction with a gravel augmentation program. By restoring access to the sediment stored in the subsurface, removal of a coarse surface layer can reduce the scale of required gravel augmentation.

4.4.7.3 Water temperature compliance point

Fall-run Chinook salmon juvenile smolt during early spring, prior to increases in water temperatures, and spawn during the fall while temperatures are declining. As a result, management of the water temperature compliance point has little potential to affect the fall run.

4.4.7.4 Spring flows to inundate shallow water habitats

Research conducted in the Central Valley suggests that seasonally inundated, shallow water habitats may provide superior rearing habitat than mainstem channel habits for juvenile salmonids (Sommer et al. 2001a). Juvenile fall-run salmon migrate downstream (February-April) during periods when floodplains and bypasses are flood periodically during wet water years. By promoting faster growth, these periods of prolonged floodplain inundation likely help the fall-run population by increasing juvenile salmon survival. Any measures that can be implemented to promote more frequent floodplain inundation during the winter and spring (e.g., flow management, diversion structures) in bypasses along the Sacramento River will likely contribute to stronger fall-run escapements.

It may also be possible to increase survival of juveniles prior to smolting by using spring pulse flows to re-connect shallow water rearing habitats within the bankfull channel with the mainstem. By maintaining shallow water rearing habitats within the bankfull channel, stranding risk would be reduced, and beneficial rearing habitat would be increased.

4.5 Late-Fall-Run Chinook Salmon

As described in the previous chapter, NMFS classifies late-fall-run Chinook salmon as part of the Central Valley Fall-run and Late-fall Chinook salmon ESU, reasoning that the late-fall-run population represents a life history variation of the fall-run salmon population rather than a distinct run (NMFS 2004c). However, agencies generally treat late-fall-run salmon in the Sacramento River basin as a distinct run, by conducting separate carcass and redd surveys for the run, and by publishing separate reports to address the fall-run and late-fall-run populations. Agencies also manage the hatchery propagation of late-fall-run Chinook separately from fall-run salmon. Except for hatchery propagation, there are relatively few restoration and management activities that focus specifically on late-fall-run Chinook in the Sacramento River, relative to the other runs of Chinook in the basin (USFWS 1996). The USFWS's *Recovery plan for the Sacramento-San Joaquin Delta Native Fishes* (1996) suggests that the lack of direct management intervention is a function of gaps in knowledge about late-fall-run Chinook in the basin, though the lack of targeted measures may also derive from the confused status of late-fall-run as a distinct run.

Late-fall-run salmon tend to be the largest individuals of the Chinook species that occur in the Sacramento River basin (USFWS 1996). Despite their large size, the run seems to be less a focus of recreational angling (Cramer and Demko 1997), probably because they migrate upstream and spawn during the rainy season when there are fewer people on the river. Fishing guides that operate in the Sacramento River are beginning to tout the "underpublicized" late-fall-run salmon fishery (http://www.mikebogue.com/salmon.htm).

In 1999, NMFS determined that listing of the Central Valley Fall and Late-fall Chinook ESU was not warranted, but considered them a candidate species (NMFS 1999a). NMFS designated the Central Valley Fall and Late Fall-run ESU as a species of concern in 2004 (NMFS 2004c).

4.5.1 Distribution

Little is known about the historical distribution of late-fall-run salmon in the Sacramento River valley. Yoshiyama et al. (1996) gleaned incidental references to late-fall-run fish from historical documents to suggest that late-fall-run Chinook historically spawned in the mainstem reaches of the upper Sacramento River and tributaries such as the Little Sacramento, Pit, and McCloud rivers. Because a significant fraction of late-fall-run juveniles oversummer in natal streams before emigrating, mainstem reaches close to coldwater sources were likely the most important spawning areas to support historical late-fall-run Chinook production. Unfortunately, there is little historical data on water temperatures in the upper Sacramento River basin to support an analysis of the stream reaches that were likely important spawning and rearing areas for late-fall-run salmon. Yoshiyama et al. (1996) also suggested the presence of historical spawning populations of late-fall-run Chinook in the American and San Joaquin Rivers prior to the era of large dam construction.

Currently, the largest spawning population of late-fall-run salmon occurs in the mainstem Sacramento River, generally above RBDD (RM 263.5). Spawning populations of late-fall-run salmon also occur in several different tributaries of the Sacramento River, including Battle, Cottonwood, Clear and Mill creeks, and the Feather and Yuba rivers (USFWS 1996). However, the sizes of these spawning populations are relatively small, with the exception of Battle Creek where late-fall-run Chinook are artificially propagated at the Coleman National Fish Hatchery (CNFH). Since 1974, CNFH has released between 200,000 and 2.5 million yearling late-fall-run salmon annually in the Sacramento River basin, primarily in Battle Creek (Cramer and Demko 1997). Hatchery-origin fish from Battle Creek likely stray upstream to spawn naturally in the mainstem Sacramento River, though the hatchery component of total late-fall-run salmon escapements in the Sacramento River is unknown (Cramer and Demko 1997).

Late-fall-run salmon spawning generally occurs between of Red Bluff Diversion Dam (243.5) and Keswick Dam (RM 302) (USFWS 1996), though Vogel and Marine (1991) indicate that a significant proportion of spawning can occur downstream of RBDD in some years when water temperatures are favorable downstream. The aerial redd surveys that occur during the period of late-fall-run salmon spawning must often contend with poor visibility caused by inclement weather and turbidity associated with rain events. As a consequence, the distribution of late-fall-run salmon spawning is generally more difficult to identify precisely, as compared with the other runs of Chinook salmon in the mainstem Sacramento River. Boat-based carcass surveys of late-fall run salmon also occur during periods of elevated winter flows, so that carcasses can be transported and recovered farther downstream relative to associated redd locations (Snider et al. 2000b).

4.5.2 Population trends

There is little information to indicate the historical abundance of late-fall-run salmon in the Sacramento River basin. In fact, late-fall-run salmon were first recognized by fishery agencies as a distinct run only after the construction of Red Bluff Diversion Dam in 1966, which permitted more accurate counting of upstream migrants and the timing of upstream migration (USFWS 1996). Between 1967 and 1976, late-fall-run salmon escapements averaged 22,000 adults (USFWS 1996), but between 1977 and 1985, escapements averaged only about 9,500 adults (Kano 2006) (Figure 4.5-1). Population estimates of late-fall-run salmon after 1985 are complicated by changes in RBDD gate operations, when the USBR began raising the dam gates during winter months to facilitate the upstream migration of winter-run Chinook salmon. Because the upstream migration of late-fall-run salmon overlaps with that of winter-run Chinook salmon, late-fall-run benefited from improved upstream access, but the accuracy of escapement estimates suffered (USFWS 1996). RBDD gate operations were revised again in 1994 so that gates are raised between September 15 and May 15, encompassing the entire upstream migration period of late-fall-run salmon and further compromising the calculation of escapements. Table 4.5.1 displays estimates of annual escapement for late-fall-run in the mainstem Sacramento River. Post-1985 escapement estimates are cruder because of the change in RBDD gate operations. In 1996, CDFG began conducting carcass surveys targeting late-fall-run salmon in the mainstem Sacramento River to support more accurate escapement estimates. The carcass surveys are usually conducted from late December through May of the following year; consequently, high flow and turbidity conditions can often reduce the recovery of carcasses and, therefore, the annual escapement estimates. Challenging field conditions forced the abandonment of the surveys conducted for the 1995-96 and 1996-97 spawning classes (Snider et al. 2000b). Population estimates of late-fall-run salmon derived from the 1997-98, 1998-99, and 1999-2000 surveys of the mainstem Sacramento River ranged between 6,231 and 9,717 adults, which fell below the 1967-1992 average of 14,159 fish that spawned in the mainstem river above RBDD (Snider et al. 1998b, 1999, 2000b).

Though the carcass surveys indicate that abundance of late-fall-run salmon is generally less than the long-term average, the lack of reliable escapement data for most of the past two decades prevents the identification of a clear trend in the population.

4.5.3 Life history

Adult late-fall-run Chinook migrate up the Sacramento River between mid-October and mid-April, with peak migration occurring in December (Vogel and Marine 1991) (Table 4.5-1). Adults spawn soon after reaching spawning areas between January and April. Fisher reports that peak spawning in the Sacramento River occurs in early February (1994), but the carcass surveys conducted in the late 1990s suggests peak spawning may occur in January. During some years, estimates of the number of carcasses and the temporal distribution of spawning were compromised due to high flow and turbidity conditions (Snider et al 1998b, 1999, 2000b).

Fry emerge from redds between April and June (Vogel and Marine 1991). Water temperature conditions in the lower Sacramento River allow for the survival of fry that emerge in April and begin dispersing immediately downstream. However, fry that emerge in the May and June likely experience significant mortality from elevated water temperatures in the lower Sacramento River. This suggests that a significant fraction of late-fall-run juveniles rear in the upper Sacramento River throughout the summer before emigrating in the following fall and early winter as large subyearlings (Fisher 1994). Summer rearing is made possible by the cold water releases from the Shasta-Trinity divisions of the Central Valley Project. Late-fall-run juveniles generally leave the Sacramento River by December (Vogel and Marine 1991), with peak emigration of smolts in October.

Table 4.5-1. Life history timing of late-fall-run Chinook salmon in the Sacramento River basin.

Life stage	Month																							
	Jan		Feb		Mar		Apr		May		Jun		Jul		Aug		Sept		Oct		Nov		Dec	
Adult entry into mainstem Sacramento River[1,2]	▓	░	░	░	░	░	░													░	░	░	▓	▓
Migration past Red Bluff Diversion Dam[1,2,3]	░	░	░	░	░	░	░													░	░	░	▓	▓
Adult holding[4]																				░	░	░	░	░
Spawning[1,2,3,5]																							░	░
Incubation	░	░	░	░	░	░	░	░	░	░	░	░												
Fry emergence[1,3]							▓	▓	▓	▓	░	░												
Stream residency[1,3]	░	░	░	░	░	░	░	░	░	░	░	░	░	░	░	░	░	░	░	░	░	░	░	░
Fry outmigration past Red Bluff Diversion Dam[2]							▓	▓	▓	▓	░	░												
Smolt outmigration past Red Bluff Diversion Dam[2]													░	░	░	░	░	░	▓	▓	░	░		
Smolt outmigration[1]	░	░	░	░	░	░	░	░	░	░											░	░	░	░
Ocean entry[3]	░	░	░	░	░	░	░	░	░	░									░	░	░	░	░	░

Sources: [1] Yoshiyama et al. 1998; [2] Cramer and Demko 1997; 3 Fisher 1994; [4] Moyle 2002; [5]Snider et al. 1998b, 1999, 2000b.

░	Period of activity
▓	Period of peak activity

4.5.4 Specific habitat requirements

General habitat requirements for Chinook salmon are described in Section 4.1. Only habitat requirements specific to late-fall-run Chinook salmon are described here.

4.5.4.1 Spawning habitat

Late-fall-run Chinook spawn primarily in the mainstem Sacramento River between Red Bluff (RM 243.5) and Keswick Dam (RM 302). CDFG conducts aerial redd surveys that target the late-fall-run spawning period, and an analysis of the surveys suggest that adults generally spawn upstream of Red Bluff Diversion Dam (RM 243.5). We hypothesize that the downstream limit of late-fall-run spawning is dictated by the summer water temperature regime. As described above, the timing of late-fall-run spawning in January through March means that fry emerge between April and June. Water temperatures in the lower Sacramento River are often too high in May and June to support fry survival, so later-emerging fry that migrate downstream likely suffer high rates of mortality and contribute little to the population. Because of the limited swimming ability of emergent fry, only those redds constructed in reaches with suitable summer water temperatures will produce juveniles that will survive and contribute to the population. We hypothesize that consistently low survival of the progeny of fish that spawn in reaches too far downstream exerted selective pressure over time, so that individuals of the population now spawn where summer water temperatures can support oversummering of juveniles.

We also hypothesize that the relatively large size of late-fall-run Chinook, coupled with their time of spawning, may permit them to spawn in areas unavailable to the other runs of Chinook that occur in the Sacramento River basin. Chapter 3 described how the bed of the upper Sacramento River has become coarser in response to the reduced sediment supply caused by Shasta Dam. Once a substantial portion of the bed surface is covered with coarse particles that cannot be mobilized, then the area is effectively armored and unavailable for spawning. However, the largest particle size that a female can mobilize in the process of redd construction is influenced by the body size and the resultant tractive force that can be applied to the bed surface. Their larger size may permit late-fall-run adults to mobilize coarser particles that cannot be mobilized by other Chinook runs, thereby granting late-fall-run females access to small areas of bed surface unavailable to other runs. Late-fall-run adults also spawn during periods when winter flow events can increase velocities, thereby helping to transport larger sediment particles once they are mobilized by the process of redd construction. Mean monthly discharge in January and February ranges between approximately 18,000 and 23,000 cfs in the upper Sacramento River (as measured at Bend Bridge, USGS Gauge No. 11377100). In contrast, flows during the peak of fall-run spawning range from 6,000 cfs to 8,000 cfs, and flows during the peak of winter-run spawning are usually around 12,000 cfs. However, any marginal increase in the spawning habitat available to late-fall-run salmon is likely offset by the increased risk of egg mortality from redd scour during high flow events in the winter. Because late-fall-run adults may spawn during periods of high flow, redds constructed on channel margins may also be vulnerable to subsequent redd desiccation.

4.5.4.2 Rearing habitat

As described in previous sections, the life history timing of late-fall-run Chinook salmon in the Sacramento River necessitates that a substantial fraction of juveniles spend a summer rearing in the upper reaches of the river to avoid lethal water temperatures in the lower Sacramento River in the late spring and early summer. These juveniles then emigrate as subyearlings when water temperatures decline in the subsequent fall, generally leaving the Sacramento River between October and December. We hypothesize that the downstream limit of late-fall-run spawning also marks the downstream limit where summer water temperatures are suitable to support juvenile rearing. As a result, we propose that the key rearing reach for juvenile late-fall-run salmon is between Keswick Dam (RM 302) and RBDD (RM 243.5).

Fry that emerge in April and immediately begin to disperse may be able to use rearing habitats farther downstream in some years if water temperature conditions are favorable; however, we hypothesize that this fraction of late-fall-run that emigrates as fry contributes little to the population. Even if fry migrating downstream in April do not experience direct mortality from water temperatures, the warmer water temperatures in the late spring may contribute indirectly to mortality by increasing the range and feeding activity of predators, especially non-native centrarchids. Relative to other juvenile emigrants that emerged earlier in the spring but migrate downstream in April (e.g., fall-run and spring-run fry), late-fall-run fry will generally be smaller and, therefore, more vulnerable to predation. Similarly, late-fall-run fry emigrate during a period when irrigation activity increases, potentially increasing the risk of entrainment, especially because emergent fry use habitats along channel margins where water diversions are typically located.

Late-fall-run juveniles that migrate downstream as subyearlings in the fall likely use rearing habitats in the middle and lower Sacramento River. However, rearing habitat in these downstream reaches are unlikely to be limiting to the population because the larger juveniles are able to use a wider range of habitats than the fry and parr that oversummer upstream. As a result, we propose that oversummering habitat is most likely the limiting factor for the late-fall-run salmon population in the Sacramento River.

4.5.5 Conceptual Model of Historical Population Dynamics

Historically, the summer water temperature regime in the Sacramento River was a key variable that influenced the life history timing and strategy of the different salmonids that occur in the basin. In this section, we present a conceptual model that suggests that the late-fall-run Chinook life history in the Sacramento River evolved as a function of the change in the summer water temperature regime caused by the operation of Shasta Dam. By eliminating a water temperature penalty imposed on the progeny of late-arriving fall-run adults, Shasta Dam operations facilitated the emergence of late-fall-run salmon as a distinct run.

The life history timing of fall-run Chinook in the Sacramento River represents a temporal balance that allows both adults and juveniles to avoid the stressful summer water temperatures in the system (Figure 4.5-2). As described in Chapter 4.4, fall-run Chinook salmon avoid stressful summer conditions by migrating upstream in the fall (September-November) when both air and water temperatures begin to cool. Because they arrive at spawning grounds with fully developed gonads, adult fall-run can spawn immediately (October-November), which allows their progeny to emerge in time (January-March) to emigrate from the Sacramento River as fry in the subsequent spring (February-May) before water temperatures become too high. The run timing of fall-run Chinook limits the spatial distribution of fall-run spawning to the alluvial reaches of mainstem rivers below flow-related obstacles because adults migrate upstream during periods of low fall baseflows. As a consequence, there is relatively little oversummering habitat in these mainstem reaches to support a yearling life history strategy, so fall-run juveniles must emigrate as fry before spring water temperatures become lethal. Historically, these spring water temperatures imposed a lethal penalty on the progeny of any late-arriving fall-run adults.

Coldwater releases from Shasta Dam have changed the summer water temperature regime of the upper Sacramento River, effectively eliminating the water temperature penalty that was imposed historically on late-arriving fall-run spawners by creating oversummering habitat. By supporting a yearling life history strategy, this oversummering habitat in the mainstem Sacramento River allowed the late-fall-run to emerge as a distinct run. Fall-run juveniles could continue to emigrate as fry or spend a summer growing in the river before emigrating as subyearlings.

We hypothesize that two primary factors contributed to the separation of fall-run and late-fall-run as distinct runs: hybridization and redd superimposition. As described in Section 4.3, Shasta Dam eliminated the spatial segregation of spring-run and fall-run spawning in the mainstem Sacramento River. Consequently, interbreeding likely occurred between the two runs. Similarly, the temporal deadlines used by CNFH to distinguish between fall-run and spring-run fish likely resulted in hybridization of the two runs as part of hatchery operations (Yoshiyama et al. 1998). We hypothesize that one effect of this hybridization was a shift in the run-timing of fall-run Chinook to earlier spawning (Figure 4.5-3), because spring-run spawn earlier than fall-run. In contrast, redd superimposition contributed to a shift toward later upstream migration and spawning in the fall-run. In the absence of a water temperature penalty imposed on the progeny of late-arriving adults, we would expect the run timing of fall-run to shift to later in the season. Late-arriving adults would be able to build their redds atop those of early spawners, thus inducing higher rates of egg mortality for early spawners and exerting a selective pressure for late spawning (Figure 4.5-3). Over time, the result of these two shifts in run-timing was the development of two separate and distinct peaks of upstream migration and spawning activity in the Sacramento River. The resultant differences in run timing, coupled with the different juvenile rearing strategies exhibited by fall-run (e.g., outmigration of fry soon after emergence in large pulses that swamp predators) and late-fall-run (e.g., juveniles oversummer in the river before emigrating as subyearlings at a larger size less vulnerable to predation by gape-limited fish) has lead to the development of two distinct runs.

Yoshiyama et al. (1996) suggest that spawning populations of late-fall-run salmon occurred in the Sacramento River prior to the construction of Shasta Dam, citing what are usually incidental references to late-fall-run salmon in several historical documents. Though these historical accounts indicate the occurrence of salmon migrating upstream and spawning in December or later on several different Central Valley tributaries, it is not clear if such migration and spawning activity occurred consistently or in substantial numbers. These historical references to late-fall-run fish may document fall-run stragglers whose progeny perished the subsequent spring and contributed little to the population, or they may indicate passage barriers that delayed the upstream migration and spawning of fall-run fish en masse.

The late-fall-run Chinook strategy is successful because a substantial fraction of juveniles oversummer in the Sacramento River before emigrating, which allows them to avoid predation through both their larger size and greater swimming ability (most fish that prey on juvenile salmon are limited to those that are small enough to swallow, or are “gape-limited,” so larger juvenile salmon can elude a certain amount of predation through size alone). One implication of this life history strategy is that rearing habitat is most likely the limiting factor for late-fall-run Chinook, especially in light of the hypothesis that the availability of oversummering habitat determines the downstream extent of spawning habitat for late-fall-run salmon, as described in Section 4.5.4.

4.5.6 Effects of Anthropogenic Changes on Late-fall-run Chinook Salmon

4.5.6.1 Effects of Shasta and Keswick dams

The previous section described how Shasta Dam operations altered summer water temperatures in the upper Sacramento River, thereby creating oversummering habitat that supported the yearling life history strategy exhibited by late-fall-run salmon and eliminated the spring temperature penalty imposed on the progeny of late-arriving spawners. This conceptual model suggests that Shasta Dam operations contributed to the emergence of late-fall-run salmon as a distinct run in the Sacramento River. An alternative conceptual model suggests that late-fall-run salmon existed prior to the construction of Shasta Dam, so that dam construction eliminated access to historical spawning habitats.

Late-fall-run salmon in the Sacramento River have been a collateral beneficiary of the operation of the Shasta and Trinity divisions of the CVP which maintain suitable water conditions for the endangered winter-run Chinook salmon. Since 1994, cold water releases designed to protect winter-run eggs incubating through the summer months have likely extended the downstream extent of suitable oversummering habitat for late-fall-run juveniles. The operation of the Shasta Temperature Control Device (TCD) since 1997 has likely contributed similar benefits by increasing the extent of oversummering habitat for late-fall-run juveniles.

4.5.6.2 Effects of Red Bluff Diversion Dam

Before 1985, RBDD gate operations likely impeded the upstream migration of late-fall-run adults. As described previously, the USBR began raising RBDD gates during winter months beginning in 1995 to facilitate the upstream passage of winter-run Chinook salmon. These changes in gate operations likely contributed to easier upstream access for late-fall-run salmon. However, a radio-tagging study conducted by CDFG between 1979 and 1981 found that late-fall-run adults (n=30) were delayed on average by 3.9 days before successfully migrating past the dam (Hallock and Fisher 1985). Such a short delay likely had little effect on the spawning success of these fish (Hallock and Fisher 1985).

4.5.6.3 Hatchery Propagation

Late-fall-run salmon have been artificially propagated at the Coleman National Fish Hatchery on Battle Creek for more than two decades. The USFWS releases between 200,000 and 2.5 million late-fall-run juveniles in the Sacramento basin each year, primarily in Battle Creek. Though hatchery strays likely compose a portion of the spawning population of late-fall-run salmon in the Sacramento River, it is unclear what proportion of escapements that hatchery-origin fish constitutes. It is also unclear if hatchery juveniles that are released in Battle Creek compete with naturally spawned juveniles for oversummering habitat in the mainstem Sacramento River.

4.5.7 Management Implications, Key Hypotheses, and Uncertainties

One of the key uncertainties about late-fall-run Chinook salmon in the Sacramento River is whether it constitutes a distinct run or whether it is a life history variation of the fall-run spawning population. Management activities currently present a confused answer, because the runs are managed separately (e.g., hatchery operations, escapement estimates, redd surveys) but NMFS considers the two runs to be part of the same ESU. We have argued that late-fall-run began as a life history variation of fall-run salmon in the Sacramento River, but changes in the water temperature regime of the upper Sacramento River now support the yearling life history strategy exhibited by late-fall-run. The combination of different run timing and different juvenile rearing strategies between the fall-run and late-fall-run salmon populations in the Sacramento River lead us to conclude that they are currently distinct runs.

4.5.7.1 Expand oversummering Habitat

The primary method for enhancing the late-fall-run salmon population in the Sacramento River would be to release more cold water from the Shasta and Trinity divisions of the CVP to maintain suitable water temperatures farther downstream, thereby expanding oversummering habitat for late-fall run juveniles. We recognize that this management measure would add to the complexity of water supply operations that resource agencies must balance, and that it could pose a conflict with the maintenance of a coldwater pool in Shasta Reservoir which is maintained for winter-run Chinook salmon. We also recognize that dedicating coldwater resources to a fish that currently has no protected status would certainly meet resistance. Nevertheless, we are compelled to suggest this strategy as a palliative for late-fall-run salmon in the Sacramento River, because we hypothesize that oversummering habitat is the limiting factor for the population.

4.5.7.2 Conduct Water Temperature Modeling

As described in Section 4.5.4, we hypothesize that the downstream limit of late-fall-run spawning, as indicated by annual redd surveys conducted by CDFG, is dictated by the downstream extent of suitable summer water temperatures for late-fall-run juveniles that oversummer in the Sacramento River. The USBR has developed a new water temperature model in the past year that predicts water temperatures in the Sacramento River as a function of different meteorological and flow variables. Application of the model to reconstruct historical water temperature conditions in the Sacramento River would support a test of this hypothesis by comparing water temperature conditions with the location of recorded redds. This analysis could also support a better understanding of how the proposed upstream movement of the water temperature compliance point for winter-run Chinook salmon would affect the late-fall-run population.

4.6 References

Ackers, P., and W. R. White. 1973. Sediment transport: new approach and analysis. Journal of the Hydraulic Division of the American Society of Civil Engineers 99: 2041-2060.

Agrawal, A., R. S. Schick, E. P. Bjorkstedt, R. G. Szerlong, M. N. Goslin, B. C. Spence, T. H. Williams, and K. M. Burnett. 2005. Predicting the potential for historical coho, Chinook and steelhead habitat in northern California. Technical Memorandum NOAA-TM-NMFS-SWFSC-379. National Marine Fisheries Service, Santa Cruz Laboratory, Southwest Fisheries Science Center, Santa Cruz, California.

Alderdice, D. F., and F. P. J. Velsen. 1978. Relation between temperature and incubation time for eggs of Chinook salmon (*Oncorhynchus tshawytscha*). Journal of the Fisheries Research Board of Canada 35: 69-75.

Alderdice, D. F., W. P. Wickett, and J. R. Brett. 1958. Some effects of temporary exposure to low dissolved oxygen levels on Pacific salmon eggs. Journal of the Fisheries Research Board of Canada 15: 229-250.

American Fisheries Society. 1985. Petition to list the winter-run of chinook salmon in the Sacramento River of California as a threatened species. Submitted by Cay Goude of the California-Nevada Chapter of the American Fisheries Society to Dr. William Gordon, Director of the National Marine Fisheries Service. October 31, 1985.

Armour, C. L. 1990. Guidance for evaluating and recommending temperature regimes to protect fish. Instream Flow Information Paper 28. Biological Report 90 (22). U. S. Fish and Wildlife Service, National Ecology Research Center, Fort Collins, Colorado.

Bams, R. A. 1970. Evaluation of a revised hatchery method tested on pink and chum salmon fry. Journal of the Fisheries Research Board of Canada 27: 1429-1452.

Banks, J. L., L. G. Fowler, and J. W. Elliott. 1971. Effects of rearing temperature on growth, body form, and hematology of fall Chinook fingerlings. The Progressive Fish-Culturist 33: 20-26.

Banks, M. A., V. K. Rashbrook, M. J. Calavetta, C. A. Dean, and D. Hedgecock. 2000. Analysis of microsatellite DNA resolves genetic structure and diversity of chinook salmon (*Oncorhynchus tshawytscha*) in California's Central Valley. Canadian Journal of Fisheries and Aquatic Sciences 57: 915-927.

Beckman, B. R., D. A. Larsen, B. Lee-Pawlak, and W. W. Dickhoff. 1998. Relation of fish size and growth rate to migration of spring chinook salmon smolts. North American Journal of Fisheries Management 18: 537-546.

Bell, M. C. 1973. Fisheries handbook of engineering requirements and biological criteria. Contract DACW57-68-C-0086. Fisheries-Engineering Research Program, U. S. Army Corps of Engineers, North Pacific Division, Portland, Oregon.

Bell, M. C. 1986. Fisheries handbook of engineering requirements and biological criteria. Report No. NTIS AD/A167-877. Fish Passage Development and Evaluation Program, U. S. Army Corps of Engineers, North Pacific Division, Portland, Oregon.

Bell, R. 1958. Time, size, and estimated numbers of seaward migrants of Chinook salmon and steelhead trout in the Brownlee-Oxbow section of the middle Snake River. State of Idaho Department of Fish and Game, Boise.

Berggren, T. J., and M. J. Filardo. 1993. An analysis of variables influencing the migration of juvenile salmonids in the Columbia River basin. North American Journal of Fisheries Management 13: 48-63.

Bjornn, T. C. 1971. Trout and salmon movements in two Idaho streams as related to temperature, food, stream flow, cover, and population density. Transactions of the American Fisheries Society 100: 423-438.

Bjornn, T. C., and D. W. Reiser. 1991. Habitat requirements of salmonids in streams. Pages 83-138 *in* W. R. Meehan, editor. Influences of forest and rangeland management on salmonid fishes and their habitats. Special Publication No. 19. American Fisheries Society, Bethesda, Maryland.

Bradford, M. J., J. A. Grout, and S. Moodie. 2001. Ecology of juvenile chinook salmon in a small non-natal stream of the Yukon River drainage and the role of ice conditions on their distribution and survival. Canadian Journal of Zoology 79: 2043-2054.

Brandes, P. L., and J. S. McLain. 2001. Juvenile Chinook salmon abundance, distribution, and survival in the Sacramento-San Joaquin estuary. Pages 39-138 *in* R. L. Brown, editor. Contributions to the biology of Central Valley salmonids. Fish Bulletin 179: Volume 2. California Department of Fish and Game, Sacramento.

Brett, J. R. 1952. Temperature tolerance in young Pacific salmon, genus *Oncorhynchus.* Journal of the Fisheries Research Board of Canada 9: 265-323.

Brett, J. R., W. C. Clarke, and J. E. Shelbourn. 1982. Experiments on thermal requirements for growth and food conversion efficiency of juvenile Chinook salmon *Oncorhynchus tshawytscha.* Canadian Technical Report of Fisheries and Aquatic Sciences 1127. Department of Fisheries and Oceans, Fisheries Research Branch, Pacific Biological Station, Nanaimo, British Columbia.

Burner, C. J. 1951. Characteristics of spawning nests of Columbia River salmon. U. S. Fish and Wildlife Service Fishery Bulletin 52: 97-110.

California Resources Agency. 1978. Sacramento River Environmental Atlas. Prepared by the Resources Agency of the State of California for the Upper Sacramento River Task Force.

Calkins, R. D., W. F. Durand, and W. H. Rich. 1940. Report of the board of consultants on the fish problem of the upper Sacramento River. Stanford University, Stanford, California.

CDFG (California Department of Fish and Game). 1981. Re: Draft report on salmon in the Sacramento River. CDFG, Environmental Services Branch. 13 August.

CDFG. 1983. Sacramento River and tributaries bank protection and erosion control investigation--evaluation of impacts on fisheries. Final Report. CDFG, Bay-Delta Fishery Project, Sacramento.

CDFG. 1990. Status and management of spring-run chinook salmon. Prepared by CDFG, Inland Fisheries Division to California Fish and Game Commission, Sacramento.

CDFG. 1997. Central Valley anadromous fish-habitat evaluations: Sacramento and American river investigations, October 1995 through September 1996. Stream Evaluation Program, Technical Report No. 97-1. Prepared by CDFG, Environmental Services Divsion, Stream Flow and Habitat Evaluation Program for U. S. Fish and Wildlife Service, Central Valley Anadromous Fish Restoration Program.

CDFG. 1998. A status review of the spring-run Chinook salmon (*Oncorhynchus tshawytscha*) in the Sacramento River drainage. Report to the Fish and Game Commission, Candidate Species Status Report 98-01. CDFG, Sacramento.

CDFG. 2002a. Sacramento River winter-run Chinook salmon. Biennial Report 2000-2001. Prepared by CDFG, Habitat Conservation Division, Native Anadromous Fish and Watershed Branch for California Fish and Game Commission.

CDFG. 2002b. Sacramento River spring-run Chinook salmon. Annual report. Prepared by CDFG, Habitat Conservation Division, Native Anadromous Fish and Watershed Branch for Fish and Game Commission.

CDFG. 2004. Sacramento River winter-run Chinook salmon. Biennial Report 2002-2003. Prepared by CDFG, Habitat Conservation Division, Native Anadromous Fish and Watershed Branch for California Fish and Game Commission.

CDFG. n. d. Spring Chinook salmon species summary. Website http://www.dfg.ca.gov/hcpb/species/ssc/sscfish/sprngrnchnok.htm

CDWR (California Department of Water Resources). 1980. Upper Sacramento River spawning gravel study. Report. Prepared for California Department of Fish and Game by CDWR, Northern District, Red Bluff.

CDWR. 1995. Sacramento River Gravel Study—Keswick Dam to Cottonwood Creek. Report. Prepared for Stacy Cepello by CDWR, Northern District, Red Bluff.

CDWR. 2004. Effects of project operations on geomorphic processes upstream of Oroville Dam. Oroville Facilities Relicensing, FERC Project No. 2100. Draft final report SP-G1. CDWR, Northern District, Red Bluff.

Chambers, J. S., R. T. Pressey, J. R. Donaldson, and W. R. McKinley. 1954. Research relating to study of spawning grounds in natural areas. Annual Report, Contract No. DA 35026-Eng-20572. Prepared by Washington State Department of Fisheries, Olympia for U. S. Army Corps of Engineers, Fisheries-Engineering Research Program, North Pacific Division, Portland, Oregon.

Chambers, J. S., G. H. Allen, and R. T. Pressey. 1955. Research relating to study of spawning grounds in natural areas. Annual Report, Contract No. DA 35026-Eng-20572. Prepared by

Washington State Department of Fisheries, Olympia for U. S. Army Corps of Engineers, Fisheries-Engineering Research Program, North Pacific Division, Portland, Oregon.

Clarke, W. C., and J. E. Shelbourn. 1985. Growth and development of seawater adaptability by juvenile fall Chinook salmon (*Oncorhynchus tshawytscha*) in relation to temperature. Aquaculture 45: 21-31.

Clarke, W. C., R. E. Withler, and J. E. Shelbourn. 1992. Genetic control of juvenile life history pattern in Chinook salmon (*Oncorhynchus tshawytscha*). Canadian Journal of Fisheries and Aquatic Sciences 49: 2300-2306.

Coble, D. W. 1961. Influence of water exchange and dissolved oxygen in redds on survival of steelhead trout embryos. Transactions of the American Fisheries Society 90: 469-474.

Combs, B. D. 1965. Effect of temperature on the development of salmon eggs. The Progressive Fish-Culturist 27: 134-137.

Combs, B. D., and R. E. Burrows. 1957. Threshold temperatures for the normal development of chinook salmon eggs. The Progressive Fish-Culturist 19: 3-6.

Cooper, A. C. 1965. The effect of transported stream sediments on the survival of sockeye and pink salmon eggs and alevin. Bulletin 18. International Pacific Salmon Fisheries Commission, New Westminster, British Columbia, Canada.

Cramer, S. P. 1987. Abundance of Rogue River fall chinook salmon. Annual Progress Report, Fish Research Project Contract AFS-78-1. Oregon Department of Fish and Wildlife, Portland.

Cramer, S. P., and D. B. Demko. 1997. The status of late-fall and spring Chinook salmon in the Sacramento River basin regarding the Endangered Species Act. Special Report. Submitted to National Marine Fisheries Service on behalf of Association of California Water Agencies and California Urban Water Agencies. Prepared by S. P. Cramer and Associates, Inc., Gresham, Oregon.

Cui, Y. In press. The Unified Gravel-Sand (TUGS) model: simulating sediment transport and gravel/sand grain size distributions in gravel-bedded rivers. Submitted to Water Resources Research.

DeHaven, R. W. 1989. Distribution, extent, replaceability and relative values to fish and wildlife of Shaded Riverine Aquatic Cover of the lower Sacramento River, California. Part I: 1987-88 study results and recommendations. Prepared by U. S. Fish and Wildlife Service, Sacramento, California for U. S. Army Corps of Engineers, Sacramento District, Sacramento, California.

Donaldson, J. R. 1955. Experimental studies on the survival of the early stages of Chinook salmon after varying exposures to upper lethal temperatures. Master's thesis. University of Washington, Seattle.

Eddy, R. M. 1972. The influence of dissolved oxygen concentration and temperature on survival and growth of Chinook salmon embryos and fry. Master's thesis. Oregon State University, Corvallis.

EA Engineering (EA Engineering, Science, and Technology). 1992. Lower Tuolumne River spawning gravel availability and superimposition. Appendix 6 to Don Pedro Project Fisheries Studies Report (FERC Article 39, Project No. 2299). *In* Report of Turlock Irrigation District and Modesto Irrigation District Pursuant to Article 39 of the License for the Don Pedro Project, No. 2299. Vol. IV. EA Engineering, Lafayette, California.

Everest, F. H., and D. W. Chapman. 1972. Habitat selection and spatial interaction by juvenile chinook salmon and steelhead trout in two Idaho streams. Journal of the Fisheries Research Board of Canada 29: 91-100.

Fisher, F. W. 1994. Past and present status of Central Valley Chinook salmon. Conservation Biology 8: 870-873.

Fris, M. B., and R. W. DeHaven. 1993. A community-based habitat suitability index model for Shaded Riverine Aquatic Cover, selected reaches of the Sacramento River system. U. S. Fish and Wildlife Service, Sacramento Field Office, Sacramento, California.

Gallagher, S. P., and M. F. Gard. 1999. Relationship between Chinook salmon (*Oncorhynchus tshawytscha*) redd densities and PHABSIM-predicted habitat in the Merced and Lower American rivers, California. Canadian Journal of Fisheries and Aquatic Sciences 56: 570-577.

Garling, D. L., and M. Masterson. 1985. Survival of Lake Michigan chinook salmon eggs and fry incubated at three temperatures. The Progressive Fish-Culturist 47: 63-66.

Gawlik, D. E. 2002. The effects of prey availability on the numerical response of wading birds. Ecological Monographs 72: 329-346.

Gladden, J. E., and L. A. Smock. 1990. Macroinvertebrate distribution and production on the floodplains of two lowland headwater streams. Freshwater Biology 24: 533-545.

Hales, Z. L., A. Shindala, and K. H. Denson. 1970. Riverbed degradation prediction. Water Resources Research 6: 549-556.

Hallock, R. J., D. H. Fry, Jr., and D. A. LaFaunce. 1957. The use of fyke traps to estimate the runs of adult salmon and steelhead in the Sacramento River. California Fish and Game 43: 271-296.

Hallock, R. J., and D. H. Fry, Jr. 1967. Five species of salmon, *Oncorhynchus*, in the Sacramento River, California. California Fish and Game 53: 5-22.

Hallock, R. J., and F. W. Fisher. 1985. Status of the winter-run Chinook salmon, *Oncorhynchus tshawytscha*, in the Sacramento River. Anadromous Fisheries Branch Office Report. California Department of Fish and Game.

Hanson, C. H. 1990. Laboratory information on the effect of water temperature on juvenile Chinook salmon in the Sacramento and San Joaquin rivers: a literature review. San Francisco Bay/Sacramento-San Joaquin Delta, Water Quality Control Plan Hearings, WQCP-SWC Exhibit 605. Prepared by Tenera, Berkeley, for State Water Contractors, Sacramento, California.

Hanson, H. A. 1940. An investigation of fish-salvage problems in relation to Shasta Dam. Special Scientific Report No. 10. U. S. Bureau of Fisheries. http://www.estuaryarchive.org/archive/SalmonidCollection/9/.

Harvey, C. D. 1995. Juvenile spring-run chinook salmon emergence, rearing and outmigration patterns in Deer Creek and Mill Creek, Tehama County for the 1994 broodyear. California Department of Fish and Game, Redding.

Harvey-Arrison, C. 2001. Re: Accounts of winter-run Chinook salmon in Battle and Mill creeks. Internal memorandum to D. Hallock, California Department of Fish and Game, Sacramento. 19 June.

Hathaway, E. S. 1927. The relation of temperature to the quantity of food consumed by fishes. Ecology 8: 428-434.

Hausle, D. A., and D. W. Coble. 1976. Influence of sand in redds on survival and emergence of brook trout (*Salvelinus fontinalis*). Transactions of the American Fisheries Society 105: 57-63.

Hayes, J. W. 1987. Competition for spawning space between brown (*Salmo trutta*) and rainbow trout (*S. gairdneri*) in a lake inlet tributary, New Zealand. Canadian Journal of Fisheries and Aquatic Sciences 44: 40-47.

Healey, M. C. 1983. Coastwide distribution and ocean migration patterns of stream- and ocean-type Chinook salmon (*Oncorhynchus tshawytscha*). Canadian Field Naturalist 97: 427-433.

Healey, M. C. 1991. Life history of Chinook salmon (*Oncorhynchus tshawytscha*). Pages 311-393 *in* C. Groot and L. Margolis, editors. Pacific salmon life histories. University of British Columbia Press, Vancouver, British Columbia.

Healey, T. P. 1979. The effect of high temperature on the survival of Sacramento River Chinook (king) salmon, *Oncorhynchus tshawytscha,* eggs and fry. Administrative Report 79-10. California Department of Fish and Game, Anadromous Fisheries Branch.

Hedgecock, D., M. A. Banks, V. K. Rashbrook, C. A. Dean, and S. M. Blankenship. 2001. Applications of population genetics to conservation of Chinook salmon diversity in the Central Valley. Pages 45-69 in R. L. Brown, editor. Contributions to the biology of Central Valley salmonids. Fish Bulletin 179: Volume 1. California Department of Fish and Game, Sacramento.

Hedgecock, D. 2002. Microsatellite DNA for the management and protection of California's Central Valley Chinook salmon (*Oncorhynchus tshawytscha*). Final report. Amendment to agreement No. B-59638. UC Davis, Bodega Bay, California.

Heming, T. A. 1982. Effects of temperature on utilization of yolk by Chinook salmon (*Oncorhynchus tshawytscha*) eggs and alevins. Canadian Journal of Fisheries and Aquatic Sciences 39: 184-190.

Hill, K. A., and J. D. Webber. 1999. Butte Creek spring-run Chinook salmon, *Oncorhynchus tshawytscha*, juvenile outmigration and life history 1995-1998. Inland Fisheries Administrative Report No. 99-5. California Department of Fish and Game, Sacramento Valley and Central Sierra Region, Rancho Cordova.

Hillman, T. W., J. S. Griffith, and W. S. Platts. 1987. Summer and winter habitat selection by juvenile Chinook salmon in a highly sedimented Idaho stream. Transactions of the American Fisheries Society 116: 185-195.

Hood, D. J. 1990. Fish abundance and distribution in the Sacramento River near Hood, California in February and March 1990. Administrative Report 59-4. California Department of Fish and Game, Inland Fisheries Division.

Jager, H. I., and K. A. Rose. 2003. Designing optimum flow patterns for fall chinook salmon in a Central Valley, California, River. North American Journal of Fisheries Management 23: 1-21.

Johnson, R., D. C. Weigand, and F. W. Fisher. 1992. Use of growth data to determine the spatial and temporal distribution of four runs of juvenile chinook salmon in the Sacramento River, California. Report No. AFF1-FRO-92-15. U. S. Fish and Wildlife Service.

Junk, W. J., P. B. Bayley, and R. E. Sparks. 1989. The flood pulse concept in river-floodplain systems. Pages 110-127 in D. P. Dodge, editor. Proceedings of the international large river symposium. Canadian Special Publication of Fisheries and Aquatic Sciences 106.

Kano, B. 2006. Grand Tab; Central Valley Streams Chinook Salmon Escapement Database. California Department of Fish and Game. Native Anadromous Fish & Watershed Branch. Red Bluff, CA.

Kelley, R. L. 1989. Battling the inland sea: Floods, public policy, and the Sacramento Valley, 1850-1986. Berkeley: University of California Press.

Kjelson, M. A., P. F. Raquel, and F. W. Fisher. 1981. Influences of freshwater inflow on chinook salmon (*Oncorhynchus tshawytscha*) in the Sacramento-San Joaquin Estuary. Pages 88-108 in R. D. Cross and D. L. Williams, editors. Proceedings of the national symposium on freshwater inflow to estuaries. FWS/OBS-81/04. U. S. Fish and Wildlife Service, Washington, D. C.

Kondolf, G. M. 2000. Assessing salmonid spawning gravel quality. Transactions of the American Fisheries Society 129: 262-281.

Kondolf, G. M., M. J. Sale, and M. G. Wolman. 1993. Modification of fluvial gravel size by spawning salmonids. Water Resources Research 29: 2265-2274.

Kondolf, G. M., and M. G. Wolman. 1993. The sizes of salmonid spawning gravels. Water Resources Research 29: 2275-2285.

Konopacky, R. C. 1984. Sedimentation and productivity in a salmonid stream. Doctoral dissertation. University of Idaho, Moscow.

Koski, K. V. 1981. The survival and quality of two stocks of chum salmon (*Oncorhynchus keta*) from egg deposition to emergence. Rapports et Proces-Verbaux des Reunions, Conseil International pour L'Exploration de la Mer 178: 330-333.

Kostow, K., editor. 1995. Biennial report on the status of wild fish in Oregon. Oregon Department of Fish and Wildlife, Portland.

Levings, C. D., and R. B. Lauzier. 1991. Extensive use of the Fraser River basin as winter habitat by juvenile Chinook salmon (*Oncorhynchus tshawytscha*). Canadian Journal of Zoology 69: 1759-1767.

Limm, M. P., and M. P. Marchetti. 2003. Contrasting patterns of juvenile Chinook salmon (*Oncorhynchus tshawytscha*) growth, diet, and prey densities in off-channel and main stem habitats on the Sacramento River. Prepared for The Nature Conservancy, Chico, California.

Lindley, S. T., R. Schick, B. P. May, J. J. Anderson, S. Greene, C. Hanson, A. Low, D. McEwan, R. B. MacFarlane, C. Swanson, and J. G. Williams. 2004. Population structure of threatened and endangered Chinook salmon ESUs in California's Central Valley Basin. Technical Memorandum NOAA-TM-NMFS-SWFSC-360. National Marine Fisheries Service, Southwest Fisheries Science Center.

Lindsay, R. B., W. J. Knox, M. W. Flesher, B. J. Smith, E. A. Olsen, and L. S. Lutz. 1986. Study of wild spring Chinook salmon in the John Day River system. 1985 Final Report, Contract DE-AI79-83BP39796, Project 79-4. Prepared by Oregon Department of Fish and Wildlife, Portland for Bonneville Power Administration, Portland, Oregon.

Lister, D. B., and H. S. Genoe. 1970. Stream habitat utilization of cohabiting underyearlings of Chinook (*Oncorhynchus tshawytscha*) and coho (*O. kisutch*) salmon in the Big Qualicum River, British Columbia. Journal of the Fisheries Research Board of Canada 27: 1215-1224.

Livesay, R. H. 1965. Channel armoring below Fort Randall Dam. Pages 461-469 in Proceedings of the Federal Inter-agency Sedimentation Conference, Jackson, Mississippi, January 28–February 1, 1963. U. S. Department of Agriculture Miscellaneous Publication No. 970.

Maragni, D. B. 2001. Chinook salmon *Oncorhynchus tshawytscha.* Pages 91-100 in Baylands ecosystem species and community profiles: life histories and environmental requirements of key plants, fish, and wildlife. San Francisco Bay Area Wetlands Ecosystem Goals Project, Oakland, California.

Marcotte, B. D. 1984. Life history, status, and habitat requirements of spring-run Chinook salmon in California. USDA Forest Service, Lassen National Forest, Chester, California.

Marine, K. R. 1997. Effects of elevated water temperature on some aspects of the physiological and ecological performance of juvenile Chinook salmon (*Oncorhynchus tshawytscha*): implications for management of California's Central Valley salmon stocks. Master's thesis. University of California, Davis.

Marine, K. R., and J. J. Cech, Jr. 2004. Effects of high water temperature on growth, smoltification, and predator avoidance in juvenile Sacramento River Chinook salmon. North American Journal of Fisheries Management 24: 198-210.

Maslin, P., M. Lennox, J. Kindopp, and W. McKinney. 1997. Intermittent streams as rearing habitat for Sacramento River chinook salmon (*Oncorhynchus tshawytscha*). Department of Biological Sciences, California State University, Chico.

McBain, S. 1989. A spawning gravel database for the McLoud River, California. Prepared for The Nature Conservancy, McCloud River Preserve, McCloud, California.

McCain, M. E. 1992. Comparison of habitat use and availability for juvenile fall Chinook salmon in a tributary of the Smith River, California. FHR Currents No. 7. USDA Forest Service, Region 5.

McCuddin, M. E. 1977. Survival of salmon and trout embryos and fry in gravel-sand mixtures. Master's thesis. University of Idaho, Moscow.

McCullough, D. A. 1999. A review and synthesis of effects of alterations to the water temperature regime on freshwater life stages of salmonids, with special reference to Chinook salmon. EPA 910-R-99-010. Prepared for U. S. Environmental Protection Agency Region 10, Seattle, Washington. www.critfc.org/tech/EPAreport.htm.

McNeil, W. J. 1964a. Effect of the spawning bed environment on reproduction of pink and chum salmon. U. S. Fish and Wildlife Service Fishery Bulletin Fishery Bulletin 65: 495-523.

McNeil, W. J. 1964b. Redd superimposition and egg capacity of pink salmon spawning beds. Journal of the Fisheries Research Board of Canada 21: 1385-1396.

McReynolds, T. R., C. E. Garman, P. D. Ward, and M. C. Schommer. 2005. Butte and Big Chico creeks spring-run chinook salmon, *Oncorhynchus tshawytscha*, life history investigation 2003-2004. Inland Fisheries Administrative Report No. 2005-1. California Department of Fish and Game, Sacramento Valley and Central Sierra Region, Rancho Cordova.

Michny, F. 1987. Sacramento River, Chico Landing to Red Bluff Project, 1986 juvenile salmon study. Prepared by U. S. Fish and Wildlife Service, Sacramento for U. S. Army Corps of Engineers, Sacramento, California.

Michny, F. 1988. Sacramento River Butte Basin reach pre-project juvenile salmon study. Prepared by U. S. Fish and Wildlife Service, Sacramento for U. S. Army Corps of Engineers, Sacramento, California.

Michny, F. 1989. Sacramento River, Chico Landing to Red Bluff Project, 1987 juvenile salmon study. Prepared by U. S. Fish and Wildlife Service, Sacramento for U. S. Army Corps of Engineers, Sacramento, California.

Michny, F., and R. Deibel. 1986. Sacramento River, Chico Landing to Red Bluff Project, 1985 juvenile salmon study. Draft report. Prepared by U. S. Fish and Wildlife Service, Sacramento, California for U. S. Army Corps of Engineers, Sacramento, California.

Michny, F., and M. Hampton. 1984. Sacramento River, Chico Landing to Red Bluff Project, 1984 juvenile salmon study. Draft report. Prepared by U. S. Fish and Wildlife Service, Sacramento, California for U. S. Army Corps of Engineers, Sacramento, California.

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

Moffett, J. W. 1949. The first four years of king salmon maintenance below Shasta Dam, Sacramento River, California. California Fish and Game 35: 77-102.

Moore, J. W., D. E. Schindler, and M. D. Scheuerell. 2004. Disturbance of freshwater habitats by anadromous salmon in Alaska. Oecologia 139: 298-308.

Moyle, P. B. 1995. Re: USFWS draft biological opinion. Letter to C. Hayashi, Office of City Attorney, City and County of San Francisco, California. 25 August.

Moyle, P. B. 2000. Abstract 89. R. L. Brown, F. H. Nichols and L. H. Smith, editors. CALFED Bay-Delta Program science conference 2000. CALFED Bay-Delta Program, Sacramento, California.

Moyle, P. B. 2002. Inland fishes of California. Revised edition. University of California Press, Berkeley.

Moyle, P. B., J. E. Williams, and E. D. Wikramanayake. 1989. Fish species of special concern of California. Final Report. Prepared by Department of Wildlife and Fisheries Biology, University of California, Davis for California Department of Fish and Game, Inland Fisheries Division, Rancho Cordova.

Moyle, P. B., R. M. Yoshiyama, J. E. Williams, and E. D. Wikramanayake. 1995. Fish species of special concern in California. Final Report. Prepared by Department of Wildlife and Fisheries Biology, University of California, Davis for California Department of Fish and Game, Inland Fisheries Division, Rancho Cordova.

Murray, C. B., and J. D. McPhail. 1988. Effect of incubation temperature on the development of five species of Pacific salmon (*Oncorhynchus*) embryos and alevins. Canadian Journal of Zoology 66: 266-273.

Murphy, M. L., J. Heifetz, J. F. Thedinga, S. W. Johnson, and K. V. Koski. 1989. Habitat utilization by juvenile Pacific salmon (*Oncorhynchus*) in the glacial Taku River, southeast Alaska. Canadian Journal of Fisheries and Aquatic Sciences 46: 1677-1685.

Myers, J. M., R. G. Kope, G. J. Bryant, D. Teel, L. J. Lierheimer, T. C. Wainwright, W. S. Grant, F. W. Waknitz, K. Neely, S. T. Lindley, and R. S. Waples. 1998. Status review of Chinook salmon from Washington, Idaho, Oregon, and California. NOAA Technical Memorandum NMFS-NWFSC-35. National Marine Fisheries Service, Northwest Fisheries Science Center, Seattle, Washington.

Myrick, C. A., and J. J. Cech, Jr. 2000. Swimming performances of four California stream fishes: temperature effects. Environmental Biology of Fishes 58: 289-295.

Myrick, C. A., and J. J. Cech, Jr. 2001. Temperature effects on chinook salmon and steelhead: a review focusing on California's Central Valley populations. Prepared by Department of Fishery and Wildlife Biology, Colorado State University, Fort Collins and Department of Wildlife, Fish, and Conservation Biology, University of California, Davis for the Bay-Delta Modeling Forum. http://www.sfei.org/modelingforum/.

Myrick, C. A. 2002. Bull trout temperature thresholds: peer review summary. Prepared for U. S. Fish and Wildlife Service, Lacey, Washington.

Myrick, C. A., D. K. Folgner, and J. J. Cech, Jr. 2004. An annular chamber for aquatic animal preference studies. Transactions of the American Fisheries Society 133: 427-433.

Needham, P. R., O. R. Smith, and H. A. Hanson. 1941. Salmon salvage problems in relation to Shasta Dam, California, and notes on the biology of the Sacramento River salmon. Transactions of the American Fisheries Society 70: 55-69.

Needham, P. R., H. A. Hanson, and L. P. Parker. 1943. Supplementary report on investigations of fish-salvage problems in relation to Shasta Dam. Special Scientific Report No. 26. U. S. Fish and Wildlife Service.

Nicholas, J. W., and D. G. Hankin. 1989. Variations in distribution of juvenile Chinook salmon in Oregon coastal river basins: implications to future stock transfer practices. Pages 182-188 *in* B. G. Shepherd, editor. Proceedings of the 1988 Northeast Pacific Chinook and coho salmon workshop. American Fisheries Society, North Pacific International Chapter, Bellingham, Washington.

Nicholas, J. W., and D. G. Hankin. 1989b. Chinook salmon populations in Oregon coastal river basins: descriptions of life histories and assessment of recent trends in run strengths. Report EM 8402. Oregon Department of Fish and Wildlife, Research and Development Section, Corvallis.

NMFS (National Marine Fisheries Service). 1989. Endangered and threatened species; critical habitat; winter-run chinook salmon. Federal Register 54: 32085-32088

NMFS. 1994. Endangered and threatened species; status of Sacramento River winter-run Chinook salmon. Federal Register 59: 440-450.

NMFS. 1997. NMFS Proposed recovery plan for the Sacramento River winter-run Chinook salmon. NMFS, Southwest Region, Long Beach, California.

NMFS. 1999a. Endangered and threatened species; threatened status for two Chinook salmon evolutionarily significant units (ESUs) in California. Federal Register 64: 50394-50415.

NMFS. 1999b. Endangered and threatened species; notice of partial 6-month extension on final listing determinations for four Evolutionarily Significant Units (ESUs) of west coast chinook salmon. Federal Register 64: 14329-14333.

NMFS. 2003. Preliminary conclusions regarding the updated status of listed ESUs of west coast salmon and steelhead. A. Chinook salmon. Draft report. Prepared by NMFS, Northwest Coast Salmon Biological Review Team, Northwest Fisheries Science Center, Seattle, Washington and Southwest Fisheries Science Center, Santa Cruz Laboratory, Santa Cruz, California.

NMFS. 2004a. Endangered and threatened species: proposed listing determinations for 27 ESUs of west coast salmonids. Federal Register 69: 33102-33179.

NMFS. 2004b. Biological opinion on the long-term Central Valley Project and State Water Project operations criteria and plan. Endangered Species Act Section 7 Consultation. NMFS, Southwest Region, Long Beach, California.

NMFS. 2004c. Endangered and threatened species: establishment of Species of Concern list, addition of species to Species of Concern list, description of factors for identifying Species of Concern, and revision of Candidate Species list Under the Endangered Species Act: notice. Federal Register 69: 19975-19979.

NMFS. 2005a. Endangered and threatened species; final listing determinations for 16 ESUs of West Coast salmon, and final 4(d) protective regulations for threatened salmonid ESUs. Federal Register 70: 37160-37204.

NMFS. 2005b. Endangered and threatened species; designation of critical habitat for 12 Evolutionarily Significant Units of west coast salmon and steelhead in Washington, Oregon, and Idaho. Federal Register 70: 52630-52858.

Pacific Fishery Management Council. 2006. Review of 2005 Ocean Salmon Fisheries. (Document prepared for the Council and its advisory entities.) Pacific Fishery Management Council, 7700 NE Ambassador Place, Suite 200, Portland, Oregon 97220-1384.

Parker, L. P., and H. A. Hanson. 1944. Experiments on transfer of adult salmon into Deer Creek, California. Journal of Wildlife Management 8: 192-198.

Pemberton, E. L. 1976. Channel changes in the Colorado River below Glen Canyon Dam. Pages 5-61 - 5-73 in Proceedings of the Federal Inter-agency Sedimentation Conference, Denver, Colorado, March 22-25, 1976. U. S. Department of Agriculture Miscellaneous Publication No. 970.

Petersen, J. H., and D. L. DeAngelis. 1992. Functional response and capture timing in an individual-based model: predation by northern squawfish (*Ptychocheilus oregonensis*) on juvenile salmonids in the Columbia River. Canadian Journal of Fisheries and Aquatic Sciences 49: 2551-2565.

Peterson, D. P., and C. J. Foote. 2000. Disturbance of small-stream habitat by spawning sockeye salmon in Alaska. Transactions of the American Fisheries Society 129: 924-934.

Phillips, R. W., R. L. Lantz, E. W. Claire, and J. R. Moring. 1975. Some effects of gravel mixtures on emergence of coho salmon and steelhead trout fry. Transactions of the American Fisheries Society 104: 461-466.

Platts, W. S., M. A. Shirazi, and D. H. Lewis. 1979. Sediment particle sizes used by salmon for spawning with methods for evaluation. Ecological Research Series EPA-600/3-79-043. U. S. Environmental Protection Agency, Corvallis Environmental Research Laboratory, Corvallis, Oregon.

Quinn, T. P. 2005. The behavior and ecology of Pacific salmon and trout. First edition. American Fisheries Society, Bethesda, Maryland and University of Washington Press, Seattle.

Reimers, P. E. 1968. Social behavior among juvenile fall Chinook salmon. Journal of the Fisheries Research Board of Canada 25: 2005-2008.

Reiser, D. W., and T. C. Bjornn. 1979. Habitat requirements of anadromous salmonids. Pages 1-54 *in* W. R. Meehan, editor. Influence of forest and rangeland management on anadromous fish habitat in western North America. General Technical Report PNW-96. USDA Forest Service, Pacific Northwest Forest and Range Experiment Station, Portland, Oregon.

Reiser, D. W., and R. T. Peacock. 1985. A technique for assessing upstream fish passage problems at small-scale hydropower developments. Pages 423-432 *in* F. W. Olson, R. G. White and R. H. Hamre, editors. Symposium on small hydropower and fisheries. American Fisheries Society, Bethesda, Maryland.

Reiser, D. W., and R. G. White. 1988. Effects of two sediment size-classes on survival of steelhead and Chinook salmon eggs. North American Journal of Fisheries Management 8: 432-437.

Reynolds, F. L., T. J. Mills, R. Benthin, and A. Low. 1993. Restoring Central Valley streams: a plan for action. California Department of Fish and Game, Inland Fisheries Division, Sacramento.

Rich, A. A. 1987. Report on studies conducted by Sacramento County to determine the temperatures which optimize growth and survival in juvenile Chinook salmon (*Oncorhynchus tshawytscha*). Prepared for McDonough, Holland and Allen, Sacramento, California by A. A. Rich and Associates, San Rafael.

Rogers, D. W. 1973. King salmon (*Oncorhynchus tshawytscha*) and silver salmon (*Oncorhynchus kisutch*) spawning escapement and spawning habitat in the upper Trinity River, 1970. Anadromous Fisheries Branch Administrative Report No. 73-10. California Department of Fish and Game, Region 1.

Roper, B. R., D. L. Scarnecchia, and T. J. La Marr. 1994. Summer distribution of and habitat use by Chinook salmon and steelhead within a major basin of the South Umpqua River, Oregon. Transactions of the American Fisheries Society 123: 298-308.

Rubin, S. P., T. C. Bjornn, and B. Dennis. 1991. Habitat suitability curves for juvenile chinook salmon and steelhead development using a habitat-oriented sampling approach. Rivers 2: 12-29.

Rutter, C. 1904. Natural history of the quinnat salmon. A report of investigations in the Sacramento River, 1896-1901. Bulletin of the United States Fish Commission 22: 65-141.

Rutter, C. 1908. The fishes of the Sacramento-San Joaquin basin, with a study of their distribution and variation. Bulletin of the U. S. Bureau of Fisheries 27: 103-152.

Silver, S. J., C. E. Warren, and P. Doudoroff. 1963. Dissolved oxygen requirements of developing steelhead trout and Chinook salmon embryos at different velocities. Transactions of the American Fisheries Society 92: 327-343.

Shirvell, C. S. 1994. Effect of changes in streamflow on the microhabitat use and movements of sympatric juvenile coho salmon (*Oncorhynchus kisutch*) and chinook salmon (*O. tshawytscha*) in a natural stream. Canadian Journal of Fisheries and Aquatic Sciences 51: 1644-1652.

Shumway, D. L., C. E. Warren, and P. Doudoroff. 1964. Influence of oxygen concentration and water movement on the growth of steelhead trout and coho salmon embryos. Transactions of the American Fisheries Society 93: 342-356.

Slater, D. W. 1963. Winter-run chinook salmon in the Sacramento River, California with notes on water temperature requirements at spawning. Special Scientific Report—Fisheries 461. U. S. Fish and Wildlife Service.

Snider, B., B. Reavis, and S. Hill. 1998a. Upper Sacramento River winter-run Chinook salmon escapement survey, April-August 1997. Stream Evaluation Program Technical Report No. 98-1. California Department of Fish and Game, Environmental Services Division.

Snider, B., B. Reavis, and S. Hill. 1998b. Upper Sacramento River late-fall-run Chinook salmon escapement survey, December 1997—May 1998. Stream Evaluation Program Technical Report No. 98-4. California Department of Fish and Game, Environmental Services Division.

Snider, B., B. Reavis, and S. Hill. 1999. Upper Sacramento River late-fall-run Chinook salmon escapement survey, December 1998—April 1999. Stream Evaluation Program Technical Report No. 99-3. California Department of Fish and Game, Habitat Conservation Division, Native Anadromous Fish and Watershed Branch.

Snider, B., B. Reavis, and S. Hill. 2000a. Upper Sacramento River fall-run Chinook salmon escapement survey, September-December 1999. Stream Evaluation Program Technical Report No. 00-3. California Department of Fish and Game, Habitat Conservation Division, Native Anadromous Fish and Watershed Branch.

Snider, B., B. Reavis, and S. Hill. 2000b. Upper Sacramento River late-fall-run Chinook salmon escapement survey, December 1999—April 2000. Stream Evaluation Program Technical Report No. 00-9. California Department of Fish and Game, Habitat Conservation Division, Native Anadromous Fish and Watershed Branch.

Snider, B., B. Reavis, and S. Hill. 2001. Upper Sacramento River winter-run Chinook salmon escapement survey, May-August 2000. Stream Evaluation Program Technical Report No. 01-1. California Department of Fish and Game, Habitat Conservation Division, Native Anadromous Fish and Watershed Branch.

Sommer, T. R., M. L. Nobriga, W. C. Harrell, W. Batham, and W. J. Kimmerer. 2001a. Floodplain rearing of juvenile Chinook salmon: evidence of enhanced growth and survival. Canadian Journal of Fisheries and Aquatic Sciences 58: 325-333.

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

Stanley, J. W. 1951. Retrogression on the lower Colorado River after 1935. American Society of Civil Engineers, Transactions 116: 943-957.

Steward, C. R., and T. C. Bjornn. 1987. The distribution of Chinook salmon juveniles in pools at three discharges. Proceedings of the Annual Conference Western Association of Fish and Wildlife Agencies 67: 364-374.

Stillwater Sciences, W. Rainey, E. Pierson, C. Corben, and M. Power. 2003. Sacramento River ecological indicators pilot study. Prepared by Stillwater Sciences, Berkeley, California for The Nature Conservancy, Chico, California.

Stuehrenberg, L. C. 1975. The effects of granitic sand on the distribution and abundance of salmonids in Idaho streams. Master's thesis. University of Idaho, Moscow.

Swales, S., R. B. Lauzier, and C. D. Levings. 1986. Winter habitat preferences of juvenile salmonids in two interior rivers in British Columbia. Canadian Journal of Zoology 64: 1506-1514.

Taylor, E. B., and P. A. Larkin. 1986. Current response and agonistic behavior in newly emerged fry of Chinook salmon, *Oncorhynchus tshawytscha*, from ocean- and stream-type populations. Canadian Journal of Fisheries and Aquatic Sciences 43: 565-573.

Thompson, K. 1972. Determining stream flows for fish life. Pages 31-50 *in* Proceedings of the instream flow requirement workshop. Pacific Northwest River Basin Commission, Vancouver, Washington.

ULEP (Umpqua Land Exchange Project). 1998. Mapping rules for Chinook salmon (*Oncorhynchus tshawytscha*). Draft Report. ULEP, Roseburg, Oregon.

Unwin, M. J. 1997. Fry-to-adult survival of natural and hatchery-produced chinook salmon (*Oncorhynchus tshawytscha*) from a common origin. Canadian Journal of Fisheries and Aquatic Science 54: 1246-1254.

USBR (U. S. Bureau of Reclamation). 2004. Long-term Central Valley Project and State Water Project Operations Criteria and Plan Biological Assessment. USDI Bureau of Reclamation, Mid-Pacific Region, Sacramento, California.

USFWS (U. S. Fish and Wildlife Service). 1995. Recovery plan for the Sacramento-San Joaquin Delta native fishes. U. S. Fish and Wildlife Service, Region 1, Portland, Oregon.

USFWS. 1996. Recovery plan for the Sacramento-San Joaquin Delta native fishes. U. S. Fish and Wildlife Service, Region 1, Portland, Oregon.

USFWS. 1999. Effect of temperature on early-life survival of Sacramento River fall- and winter-run Chinook salmon. Final report. USFWS, Northern Central Valley Fish and Wildlife Office, Red Bluff, California.

USFWS. 2003. Flow-habitat relationships for steelhead and fall, late-fall and winter-run Chinook salmon spawning in the Sacramento River between Keswick Dam and Battle Creek. Final report. USFWS, Sacramento Fish and Wildlife Office, Sacramento, California.

USFWS. 2004. Flow-habitat relationships for spring-run Chinook salmon spawning in Butte Creek. USFWS, Sacramento, California.

Van Woert, W. 1958. Time pattern of migration of salmon and steelhead into the upper Sacramento River during the 1957-1958 season. Inland Fisheries Administrative Report 58-7. California Department of Fish and Game.

Van Woert, W. F., and E. J. Smith, Jr.. 1962. Chinook salmon in Sacramento River, Battle Creek, Cottonwood Creek, Cow Creek, California. Unpublished data.

Varanasi, U. 2005. Re: Biological implications of recent ocean conditions. Memorandum from U. Varanasi, Science Director, National Marine Fisheries Service (NMFS), Northwest Fisheries Science Center, Seattle, Washington to D. R. Lohn, Regional Administrator, NMFS, Northwest Region, Seattle. 12 August.

Vaux, W. G. 1968. Intragravel flow and interchange of water in a streambed. Fishery Bulletin 66: 479-489.

Vernier, J.-M. 1969. Chronological table of embryonic development of rainbow trout. Canada Fisheries and Marine Service Translation Series 3913.

Vogel, D. A. 1987a. Estimation of the 1986 spring Chinook salmon run in Deer Creek, California. Report No. FR1/FAO-87-3. U. S. Fish and Wildlife Service.

Vogel, D. A. 1987b. Estimation of the 1986 spring Chinook salmon run in Mill Creek, California. Report No. FR1/FAO-87-12. U. S. Fish and Wildlife Service.

Vogel, D. A., and K. R. Marine. 1991. Guide to upper Sacramento River chinook salmon life history. Prepared by CH2M Hill, Redding, California for U. S. Bureau of Reclamation, Central Valley Project.

Vronskiy, B. B. 1972. Reproductive biology of the Kamchatka River chinook salmon (*Oncorhynchus tshawytscha* [Walbaum]). Journal of Ichthyology 12: 259-273.

Vyverberg, K., W. Snyder, and R. G. Titus. 1997. Lower American River chinook salmon spawning habitat evaluation, October 1994. California Department of Fish and Game, Environmental Services Division, Sacramento.

Ward, P. D., and T. R. McReynolds. 2001. Butte and Big Chico creeks spring-run Chinook salmon, *Oncorhynchus tshawytscha*, life history investigation 1998-2000. Inland Fisheries Administrative Report No. 2001-2. California Department of Fish and Game, Sacramento Valley and Central Sierra Region, Rancho Cordova.

Ward, P. D., T. R. McReynolds, and C. E. Garman. 2002. Butte and Big Chico creeks spring-run chinook salmon, *Oncorhynchus tshawytscha*, life history investigation 2000-2001. Inland Fisheries Administrative Report. California Department of Fish and Game, Sacramento Valley and Central Sierra Region, Rancho Cordova.

Ward, P. D., T. R. McReynolds, and C. E. Garman. 2003. Butte and Big Chico creeks spring-run chinook salmon, *Oncorhynchus tshawytscha*, life history investigation 2001-2002. Inland Fisheries Administrative Report. California Department of Fish and Game, Sacramento Valley and Central Sierra Region, Rancho Cordova.

Watercourse Engineering, Inc. 2002. Historic Flow and Temperature Modeling of the Sacramento River – Period of Simulation: 1970-2001. Prepared for the United States Geological Survey, Biological Resources Division, Mid-Continent Ecological Science Center, Ft. Collins, CO. March 28.

Wickett, W. P. 1954. The oxygen supply to salmon eggs in spawning beds. Journal of the Fisheries Research Board of Canada 11: 933-953.

Williams, G. P., and M. G. Wolman. 1984. Downstream effects of dams on alluvial rivers. Geological Survey Professional Paper 1286. U. S. Geological Survey, Washington, D. C.

Yoshiyama, R. M., E. R. Gerstung, F. W. Fisher, and P. B. Moyle. 1996. Historical and present distribution of chinook salmon in the Central Valley drainage of California. Pages 309-362 in Sierra Nevada Ecosystem Project: final report to congress. Volume III: Assessments, commissioned reports, and background information. University of California, Center for Water and Wildland Resources, Davis.

Yoshiyama, R. M., F. W. Fisher, and P. B. Moyle. 1998. Historical abundance and decline of Chinook salmon in the Central Valley region of California. North American Journal of Fisheries Management 18: 487-521.

Yoshiyama, R. M., E. R. Gerstung, F. W. Fisher, and P. B. Moyle. 2001. Historical and present distribution of chinook salmon in the Central Valley drainage of California. Pages 71-176 in R. L. Brown, editor. Contributions to the biology of Central Valley salmonids. Fish Bulletin 179: Volume 1. California Department of Fish and Game, Sacramento.

Yoshiyama, R. M., and P. B. Moyle. 1995. Late-fall run chinook salmon in the Central Valley. Informational memorandum to Office of the City Attorney, San Francisco, California. 6 March.

Figures

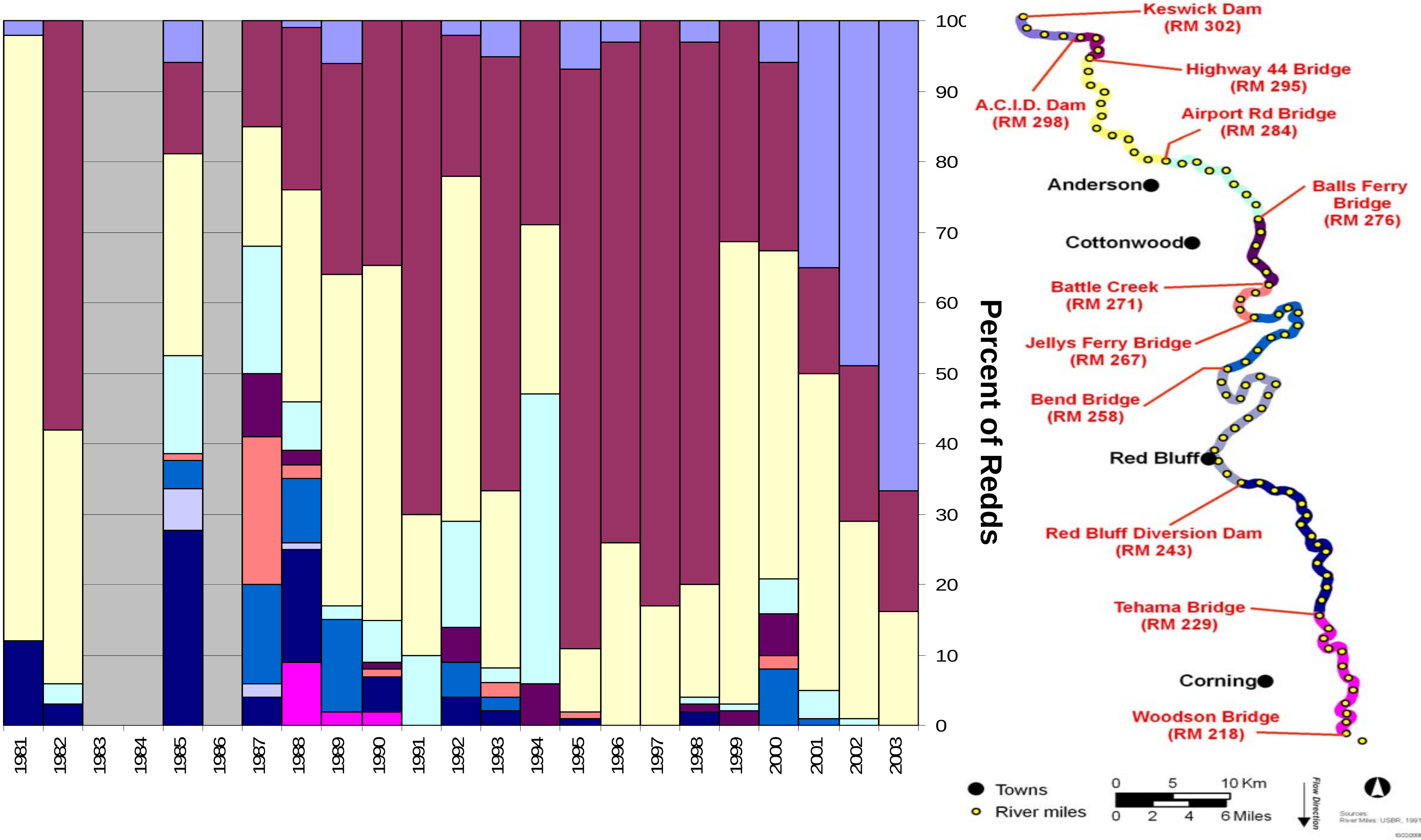
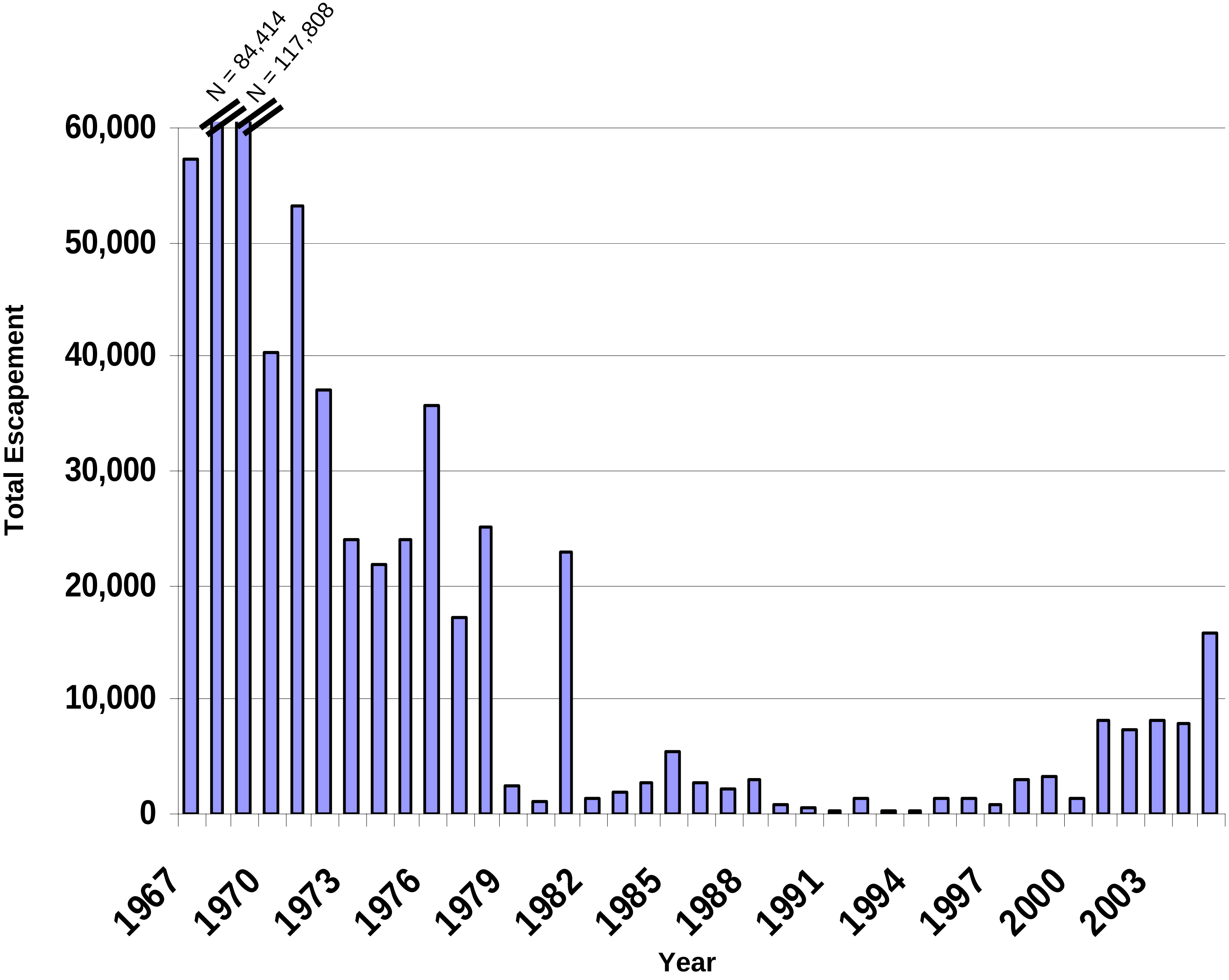


Figure 4.2-1. Redd distribution of winter-run Chinook salmon in the Sacramento River, as determined by DFG aerial redd surveys. Improvement in the fish passage facilities of ACID Dam (RM 298.4) in 2001 facilitated an upstream shift in the distribution of winter-run Chinook salmon spawning in the upper Sacramento River.



Year	Total
1967	57,306
1968	84,414
1969	117,808
1970	40,409
1971	53,089
1972	37,133
1973	24,079
1974	21,897
1975	23,930
1976	35,596
1977	17,214
1978	25,012
1979	2,364
1980	1,156
1981	22,797
1982	1,281
1983	1,831
1984	2,763
1985	5,407
1986	2,596
1987	2,185
1988	2,878
1989	696
1990	429
1991	211
1992	1,240
1993	387
1994	186
1995	1,297
1996	1,337
1997	880
1998	3,002
1999	3,288
2000	1,352
2001	8,224
2002	7,464
2003	8,218
2004*	7,869
2005*	15,839

* Preliminary data

Figure 4.2-2. Annual escapements of winter-run Chinook salmon in the Sacramento River basin (1967−2005) (Source: CDFG 2004, Grandtab.xls).

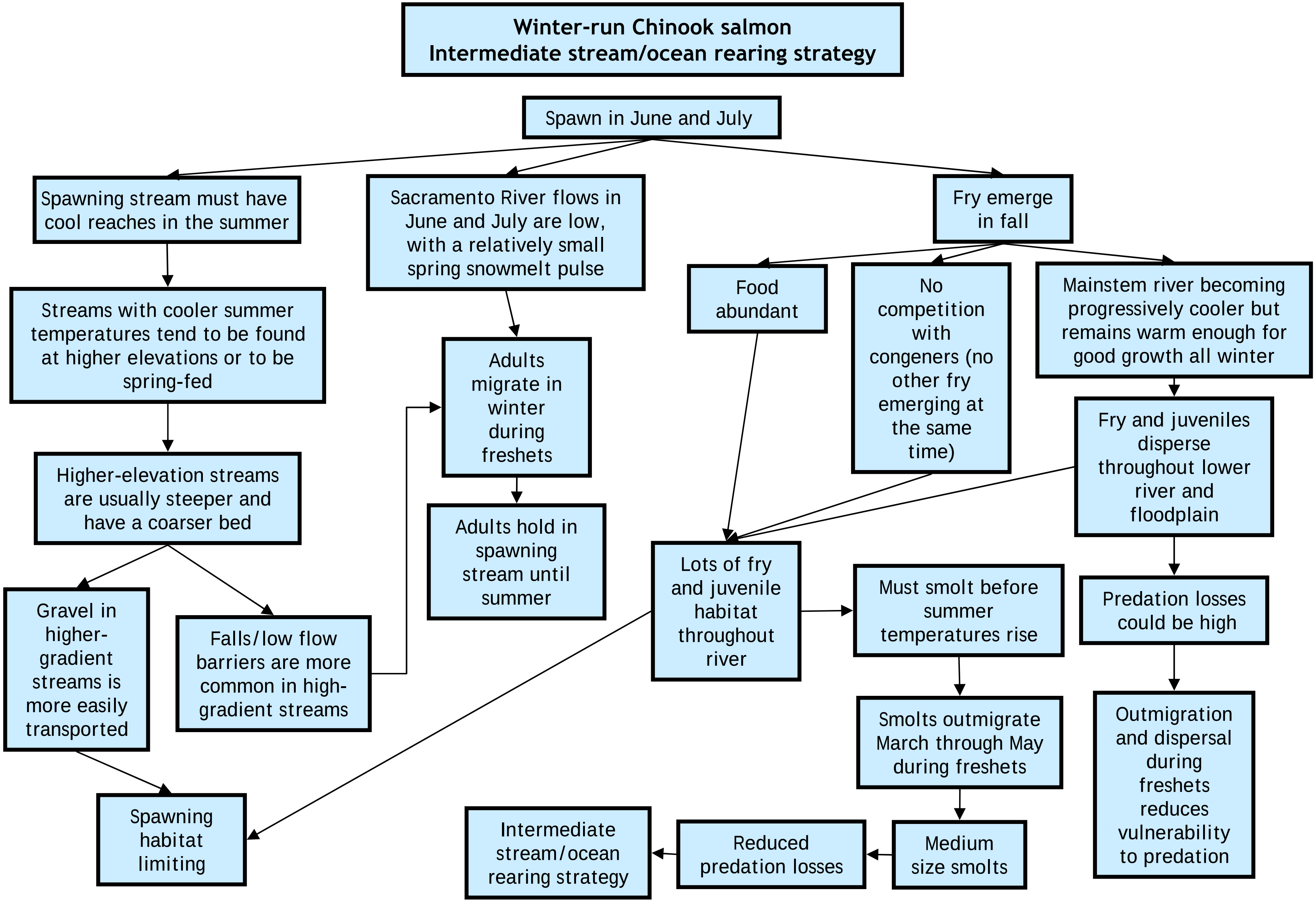


Figure 4.2-3. Winter-run Chinook salmon conceptual model of the Sacramento River.

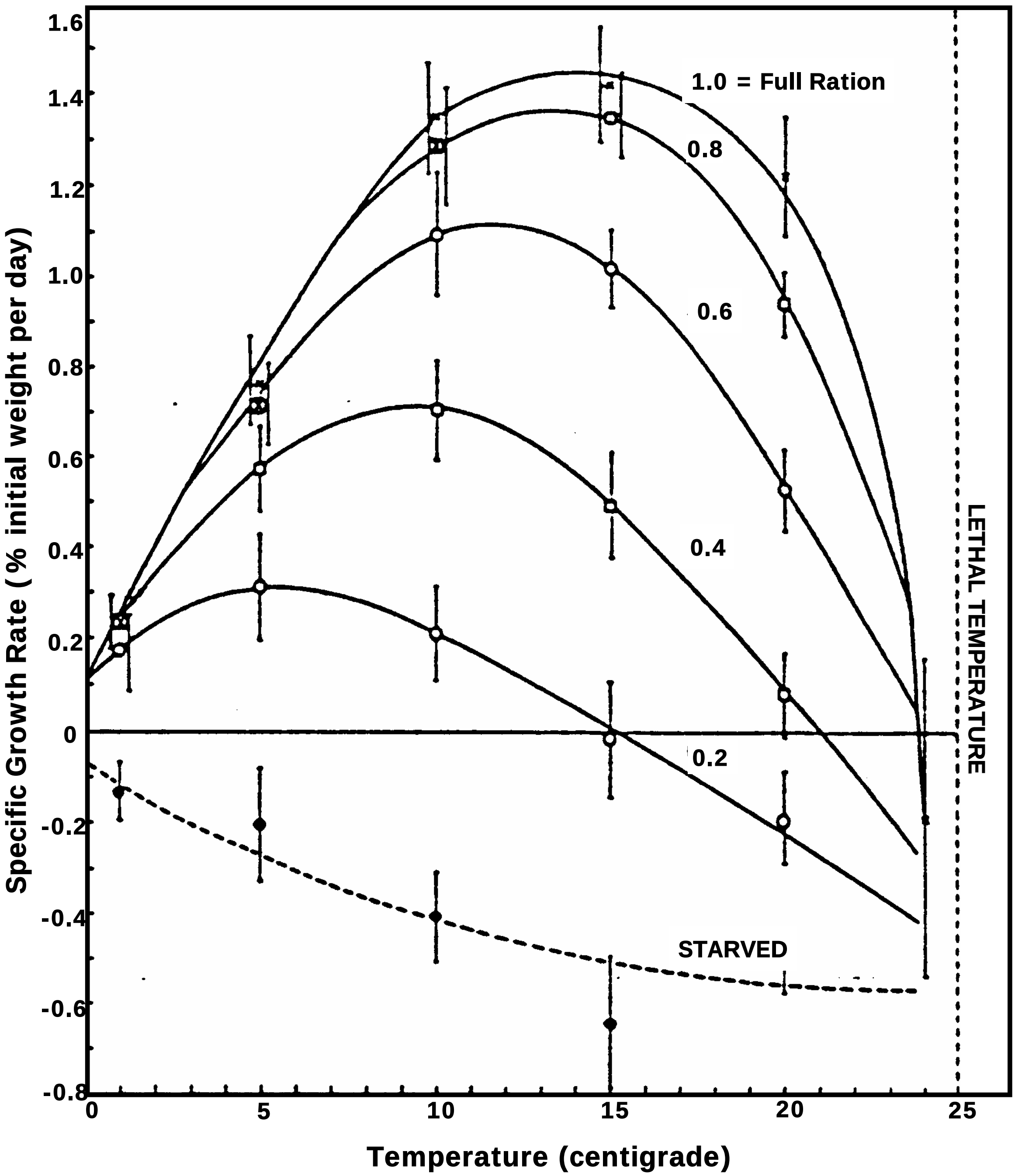


Figure 4.2-4. The effects of water temperature and food availability on juvenile sockeye salmon growth, based on studies by Brett et al. (1969). Sockeye salmon juveniles were held at a variety of temperatures at each temperature were fed different food quantities. During this laboratory experiment, increased temperatures resulted in increased growth rate up to some optimal point, beyond which growth rates declined.

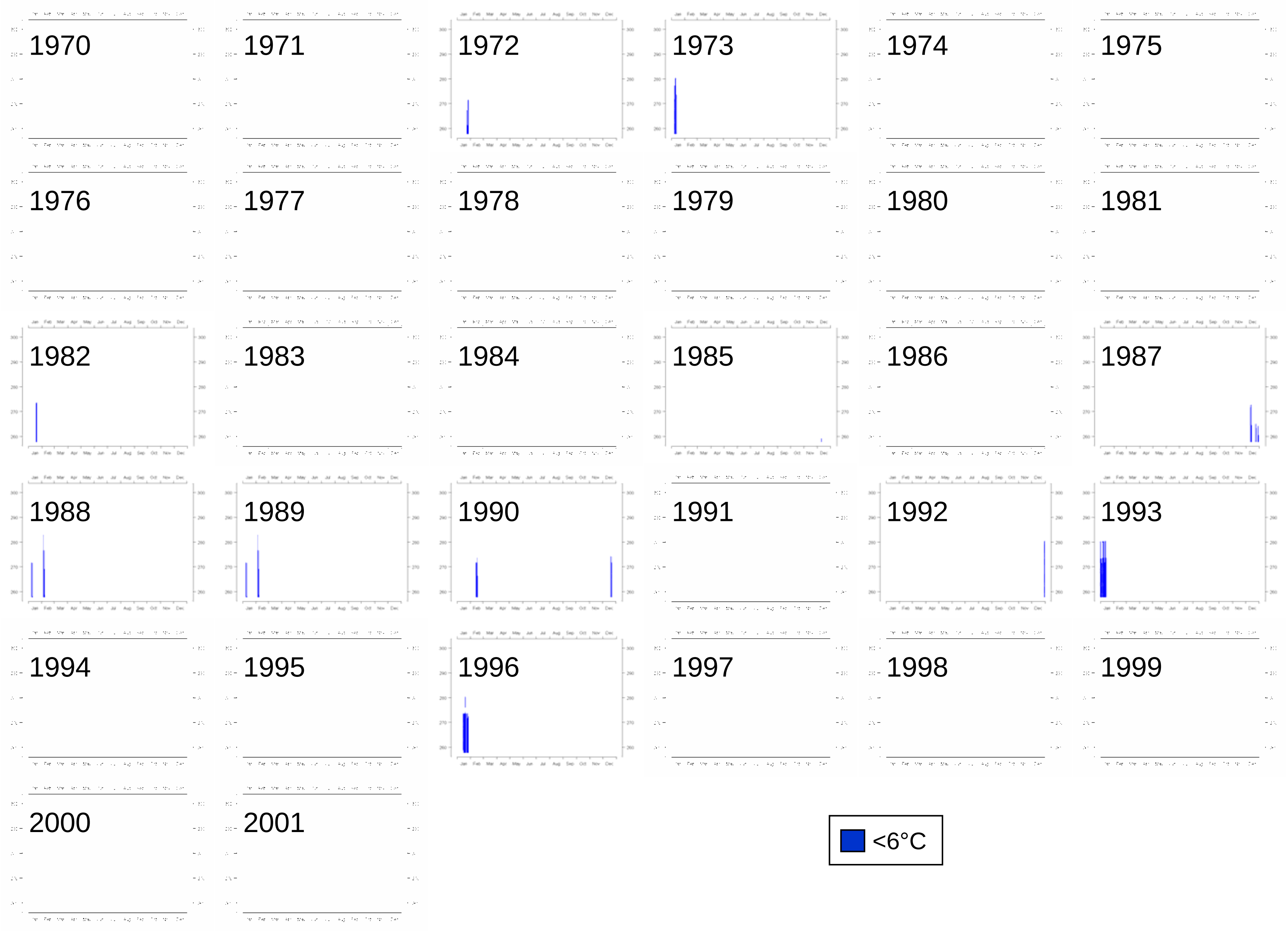


Figure 4.2-5. Days when water temperatures fell below 43˚F (6˚C), 1970-2001.

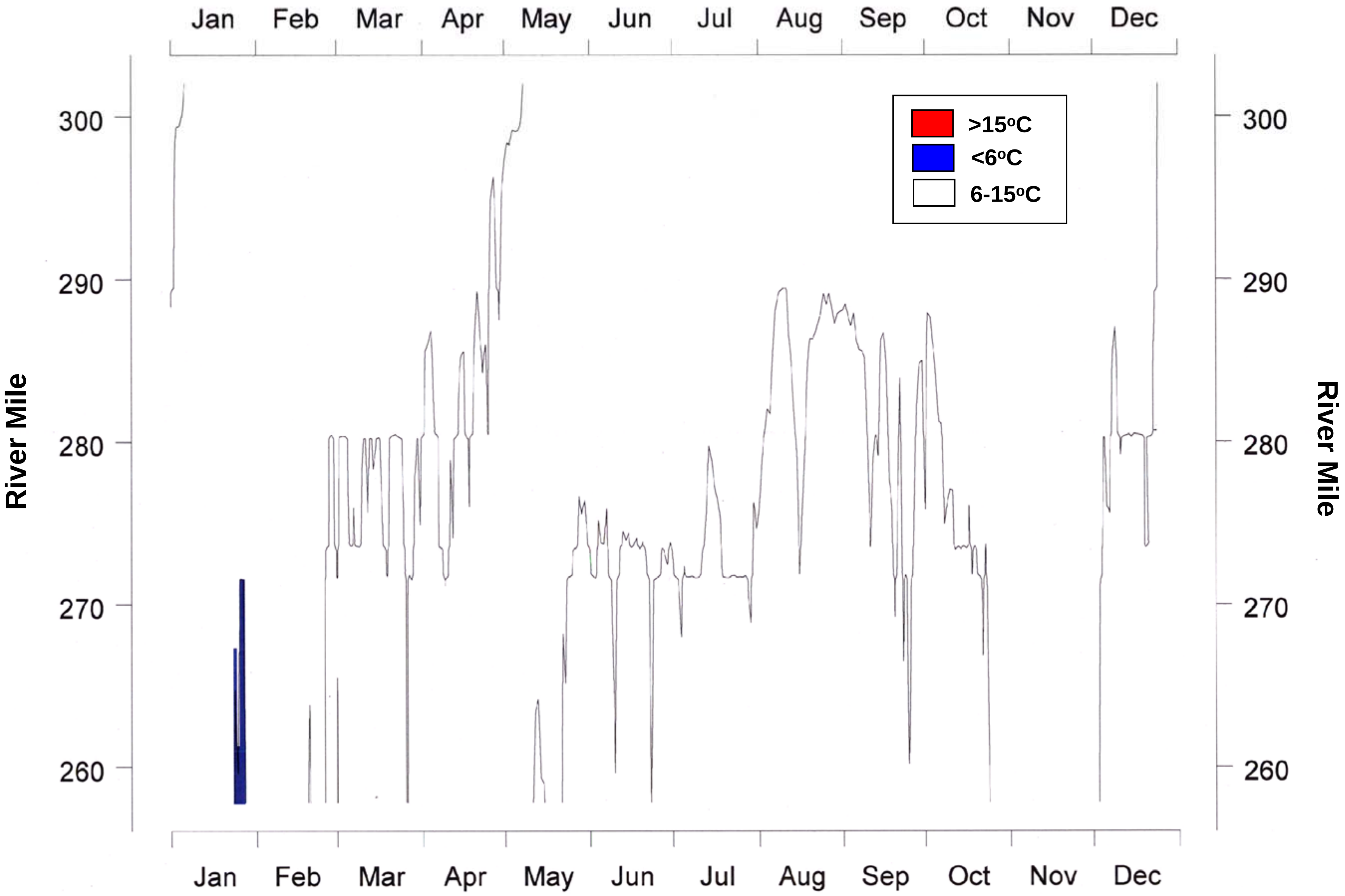


Figure 4.2-6. Example year (1972) where temperatures on the Sacramento River fall below 6°C.

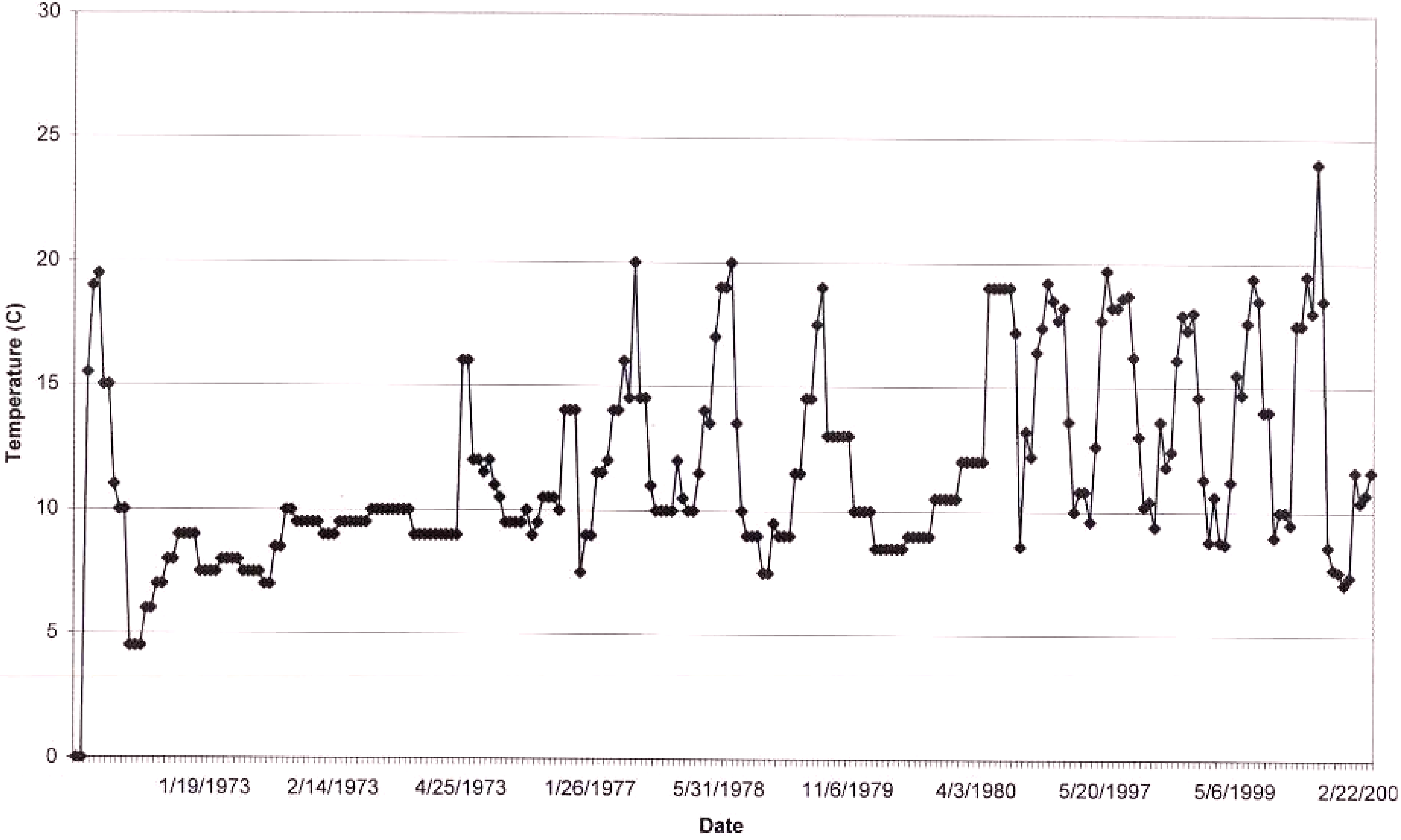


Figure 4.2-7. Temperature data collected on the Sacramento River downstream of Wilkins Slough (RM 118) between 1973 and 2000. Source data: Wilkins Slough gaging station (#11390500).

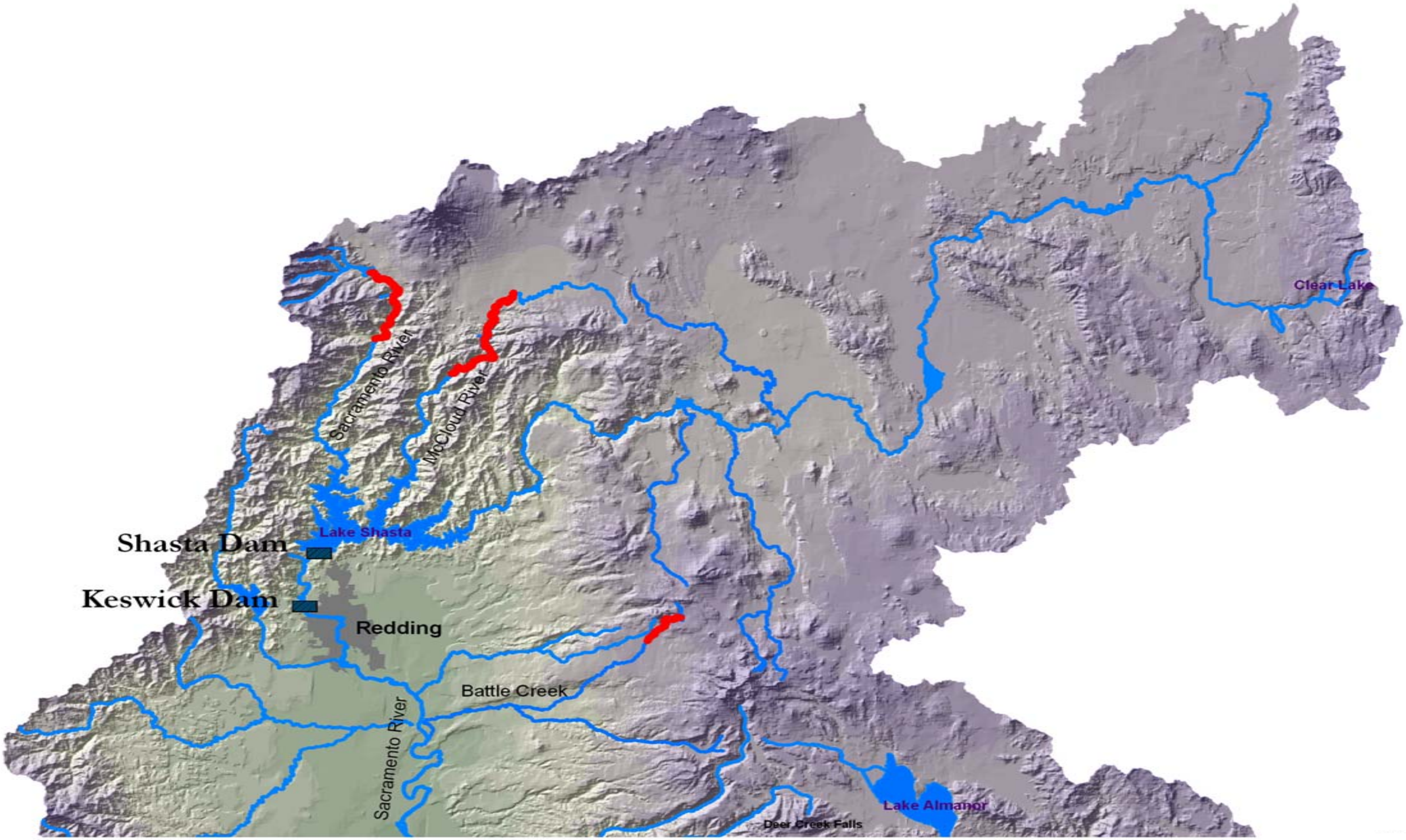


Figure 4.2-8. Potential habitat for winter run Chinook salmon upstream of Shasta Dam. Data based on Hanson 1940 and current water temperatures.

Figure 4.2-9. Habitat in the McCloud River. Stand pipes indicate spawning gravel patches.

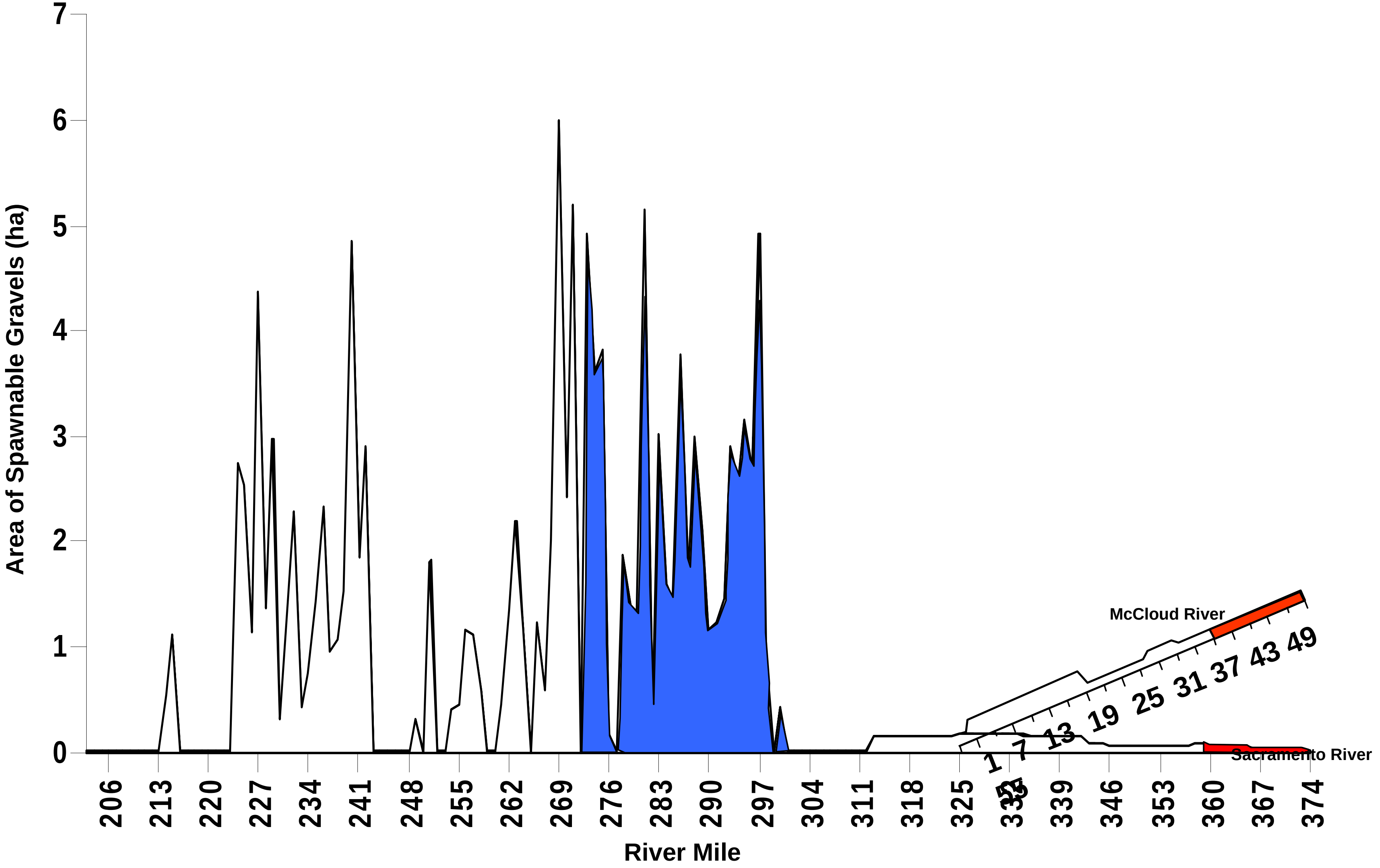


Figure 4.2-10. Spawning gravels historically available to winter-run Chinook salmon in the Sacramento and the McCloud rivers are shown in red which are based on mapped gravels by Hanson et al. 1940 and temperature suitability criterion of <16^{o}C. Spawning gravels available below Keswick in 1964 are shown in blue which are based on gravels mapped by DWR (California Resources Agency 1978) and a temperature criterion of <16^{o}C.

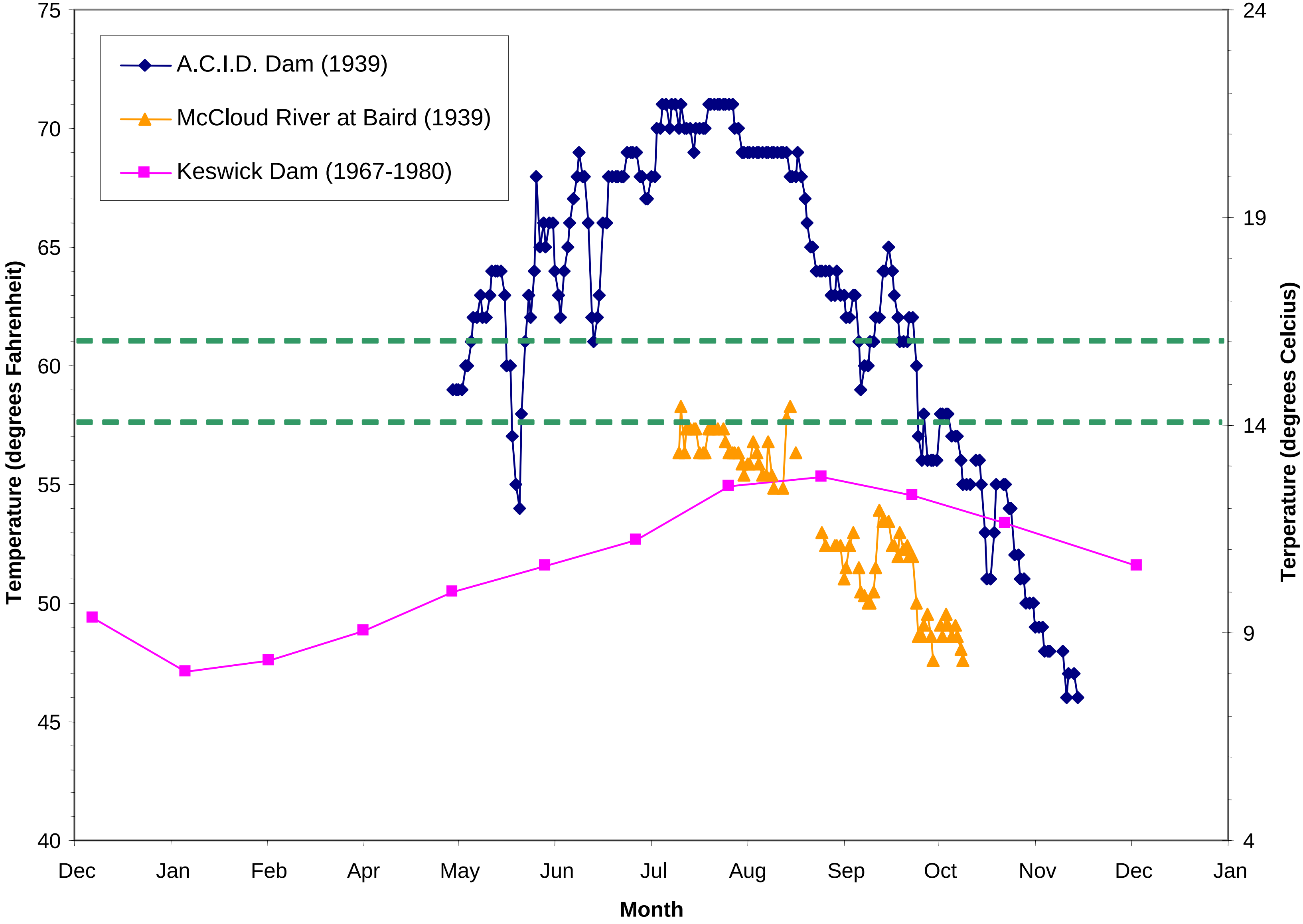


Figure 4.2-11. Temperatures at A.C.I.D Dam and the McCloud River at Baird and below Keswick Dam (1967-1980 average). Temperatures below 57ºF (14ºC) generally allow for high survival of incubating eggs and temperatures above 61ºF (16ºC) result in mortality.

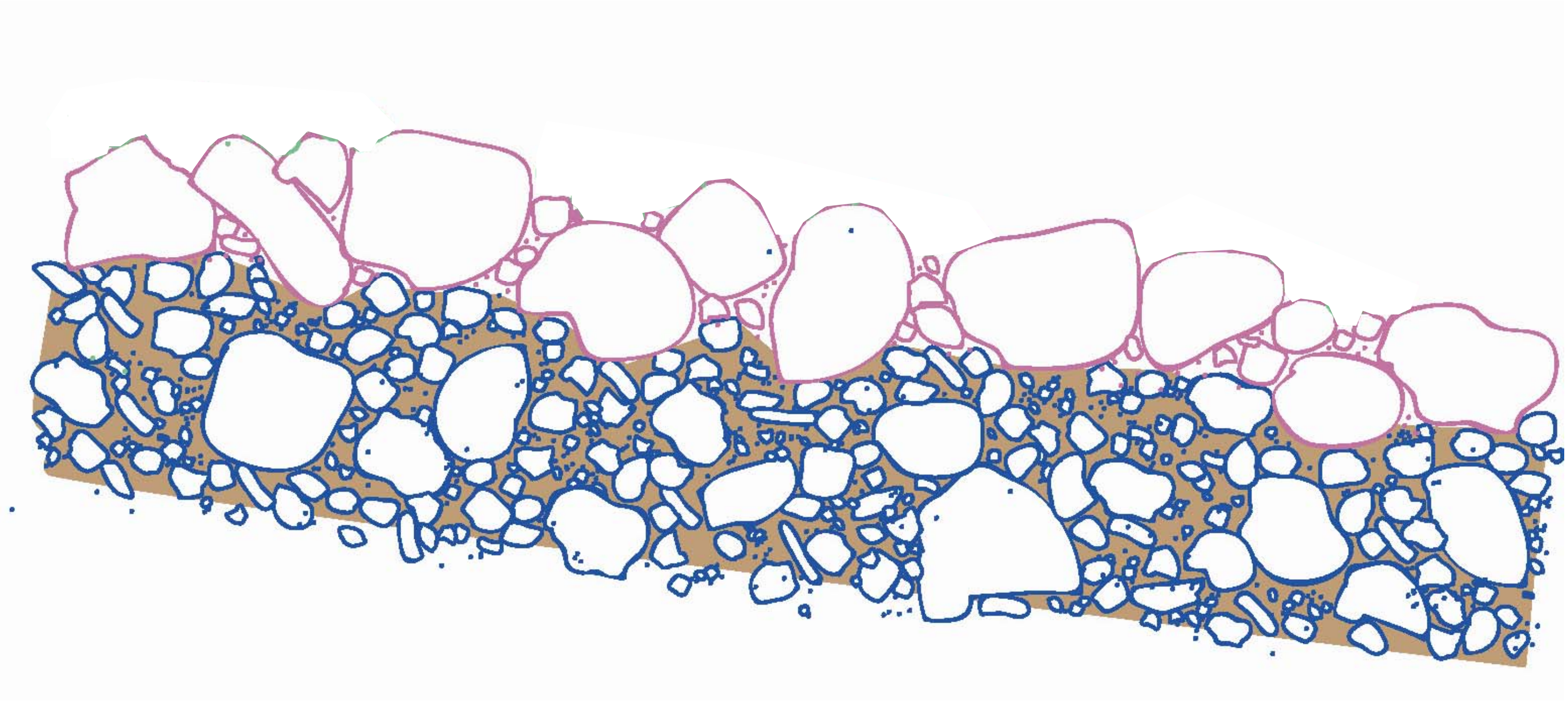


Figure 4.2-12. Illustration of an armored bed. As high flow releases from Shasta Dam transported gravels downstream, a greater proportion of the channel bed surface was covered by larger particles that could not be mobilized by high flow events. This armor layer traps gravels in the subsurface, making them unavailable for spawning.

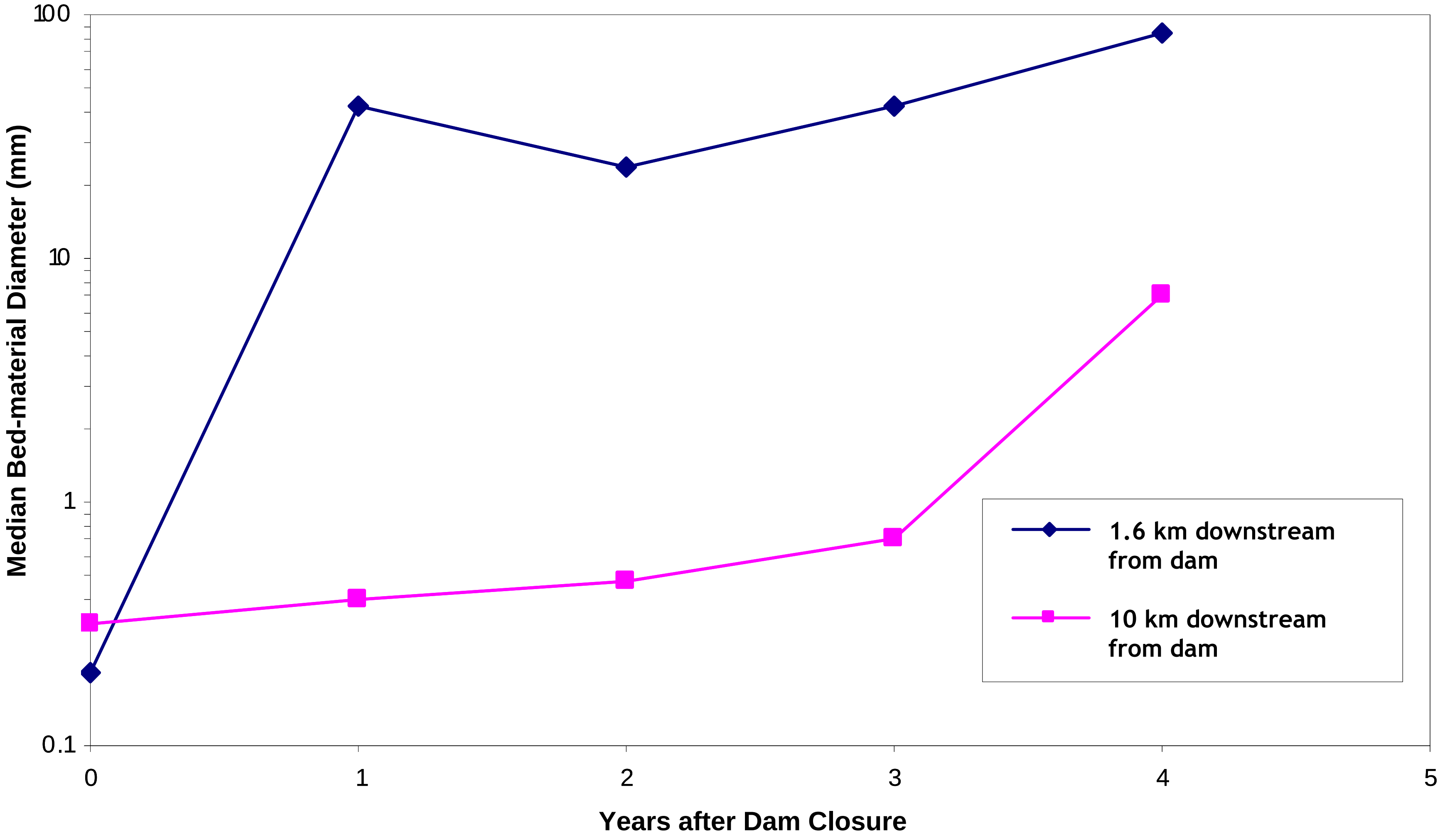


Figure 4.2-13. Downstream effects on bed grain size of Hoover Dam, Colorado River. Source: Williams and Wolman 1984.

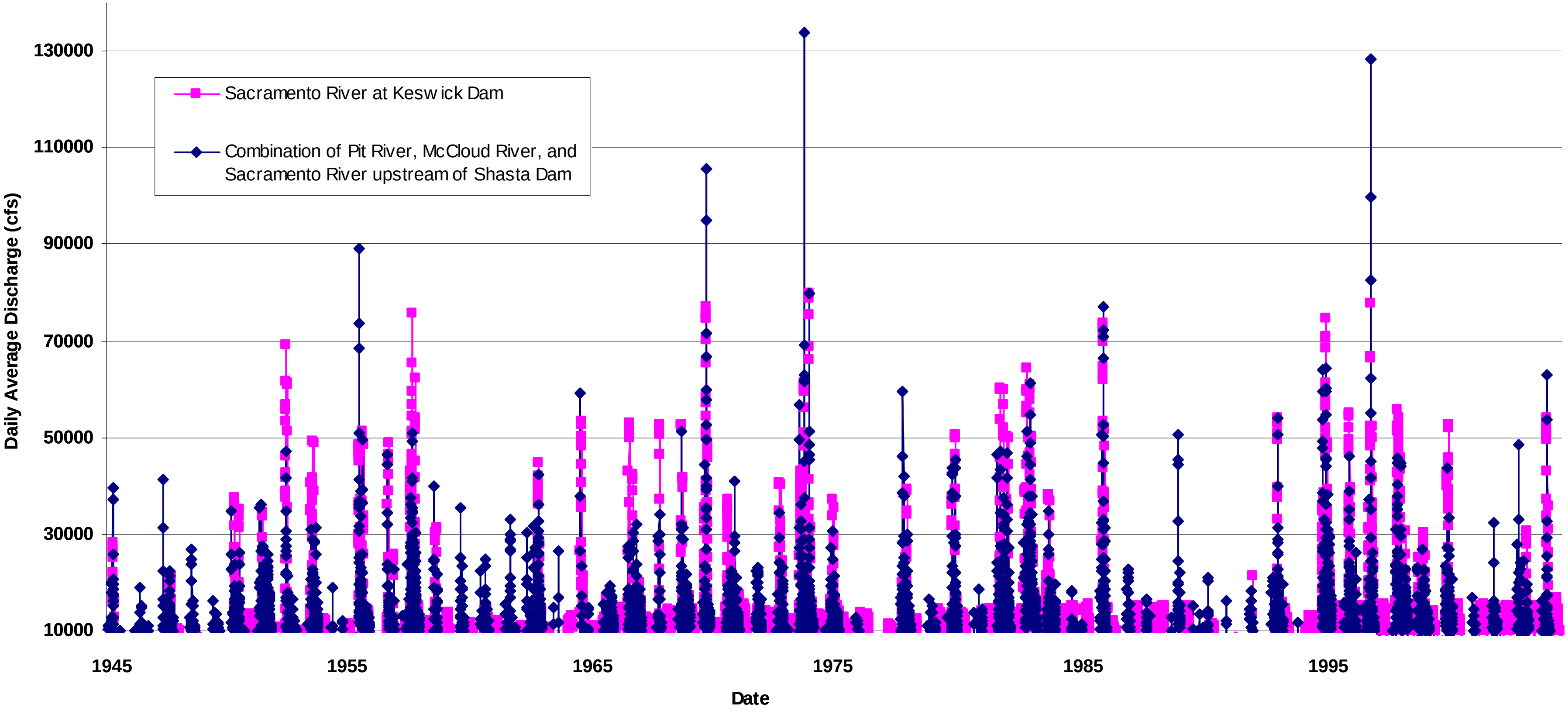


Figure 4.2-14. Daily average discharge (cfs) in the Sacramento River at Keswick Dam and in a combination of the Pit, McCloud, and Sacramento River upstream of Shasta Dam from 1945 to 2004.

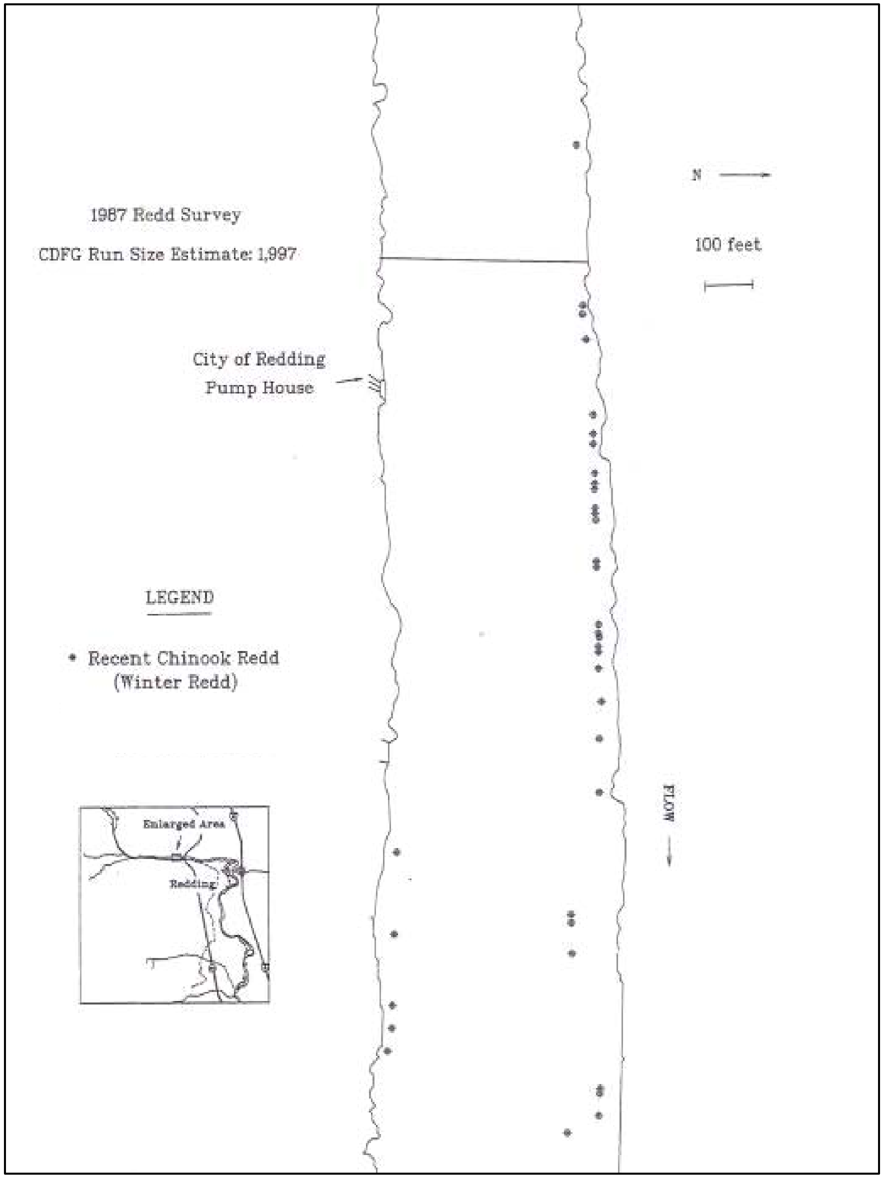


Figure 4.2-15. Winter-run Chinook salmon redd locations in the Sacramento River upstream of Diselhorst Bridge (RM 299.0 −299.3). Source: Bigelow 1996.

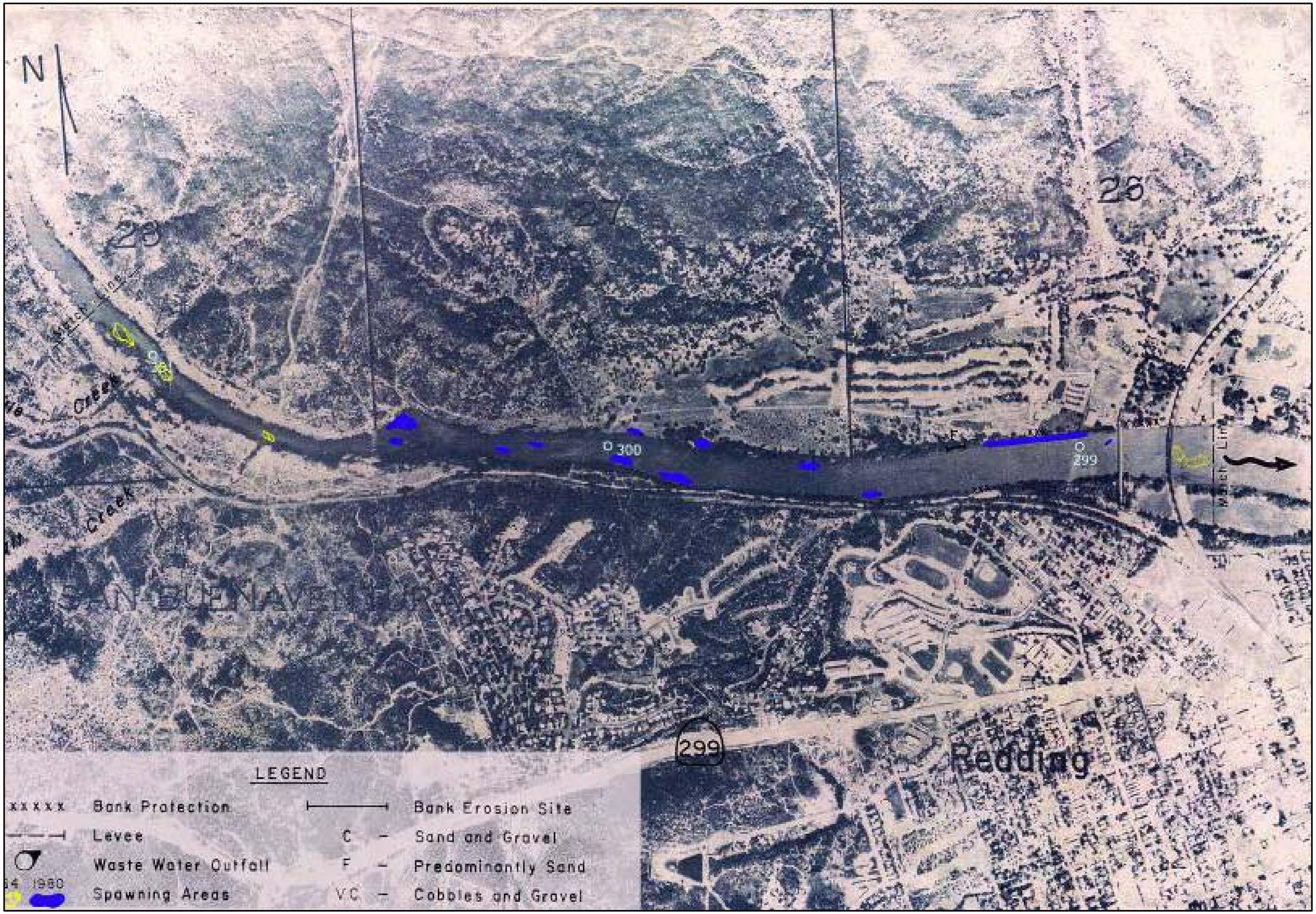


Figure 4.2-16. Spawning habitat upstream of ACID in 1964 (yellow) and 1980 (blue). Source: CDWR 1980.

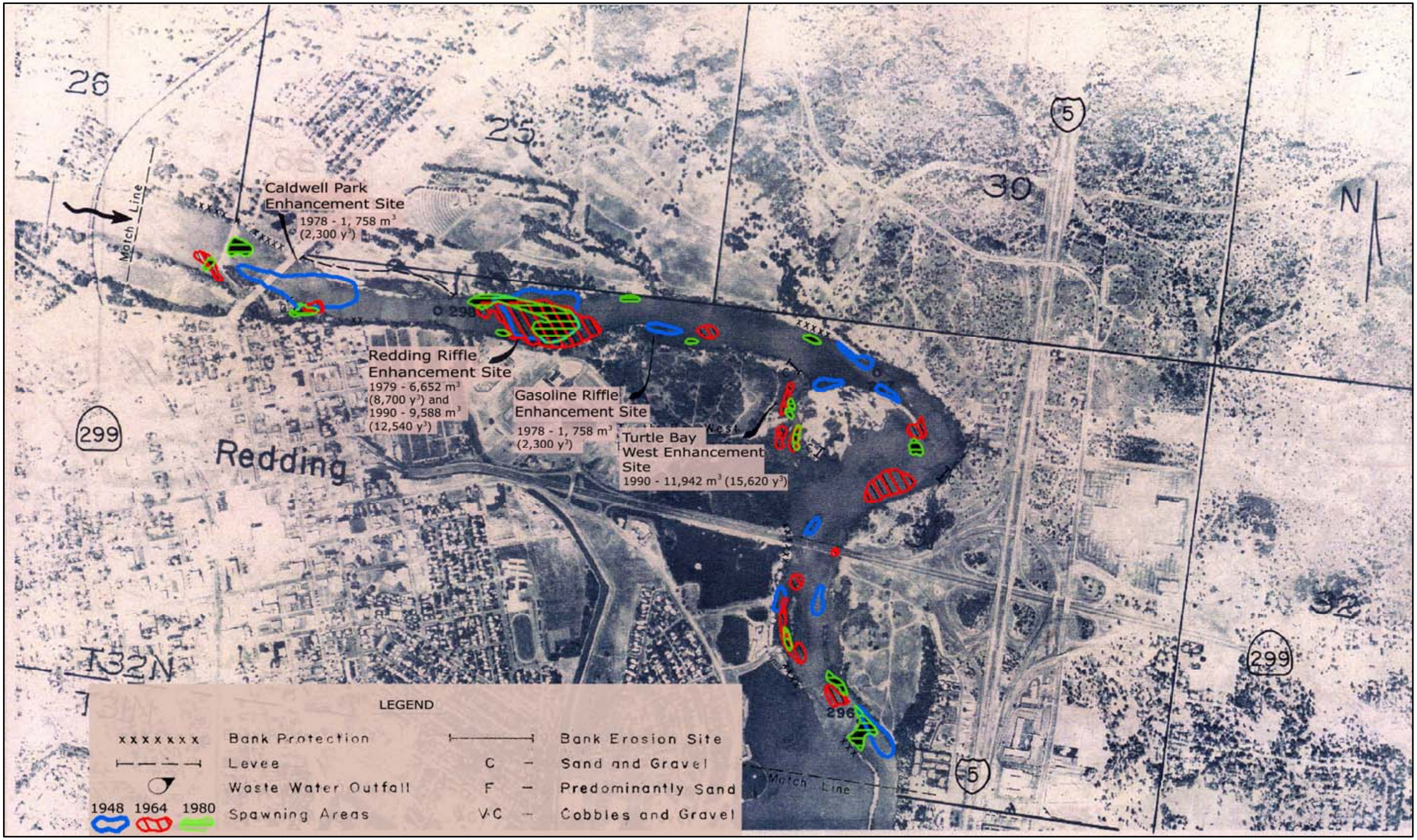


Figure 4.2-17. Spawning habitat downstream of ACID in 1964 (yellow) and 1980 (blue). The map identifies three "enhancement sites," where CDFG added ~7000 cubic yards of gravel in 1978-1979. In the photo, ACID Dam is upstream of the Caldwell Park enhancement site. Source: CDWR 1980.

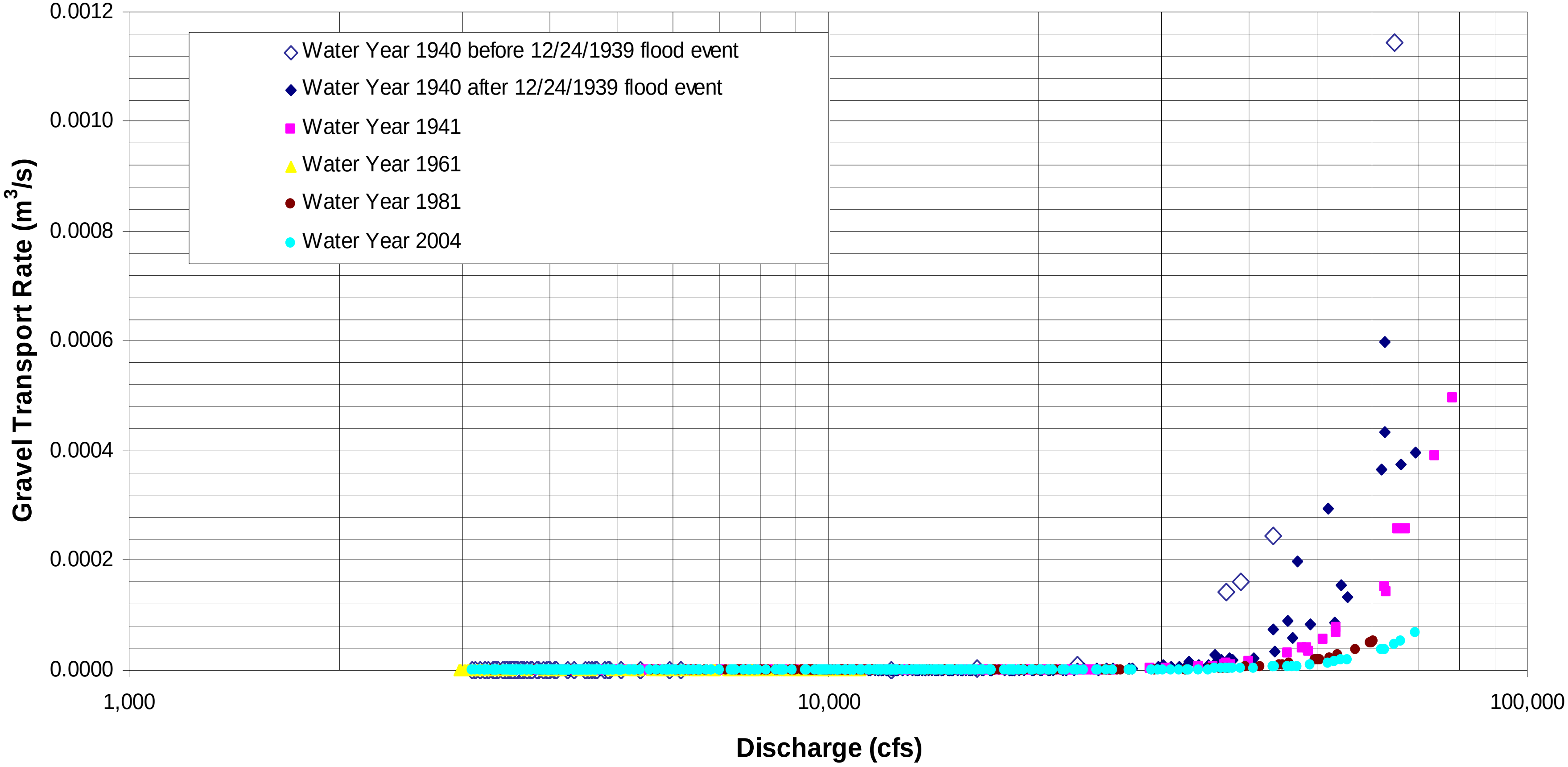


Figure 4.2-18. Gravel transport rating curve developed for RM 294.

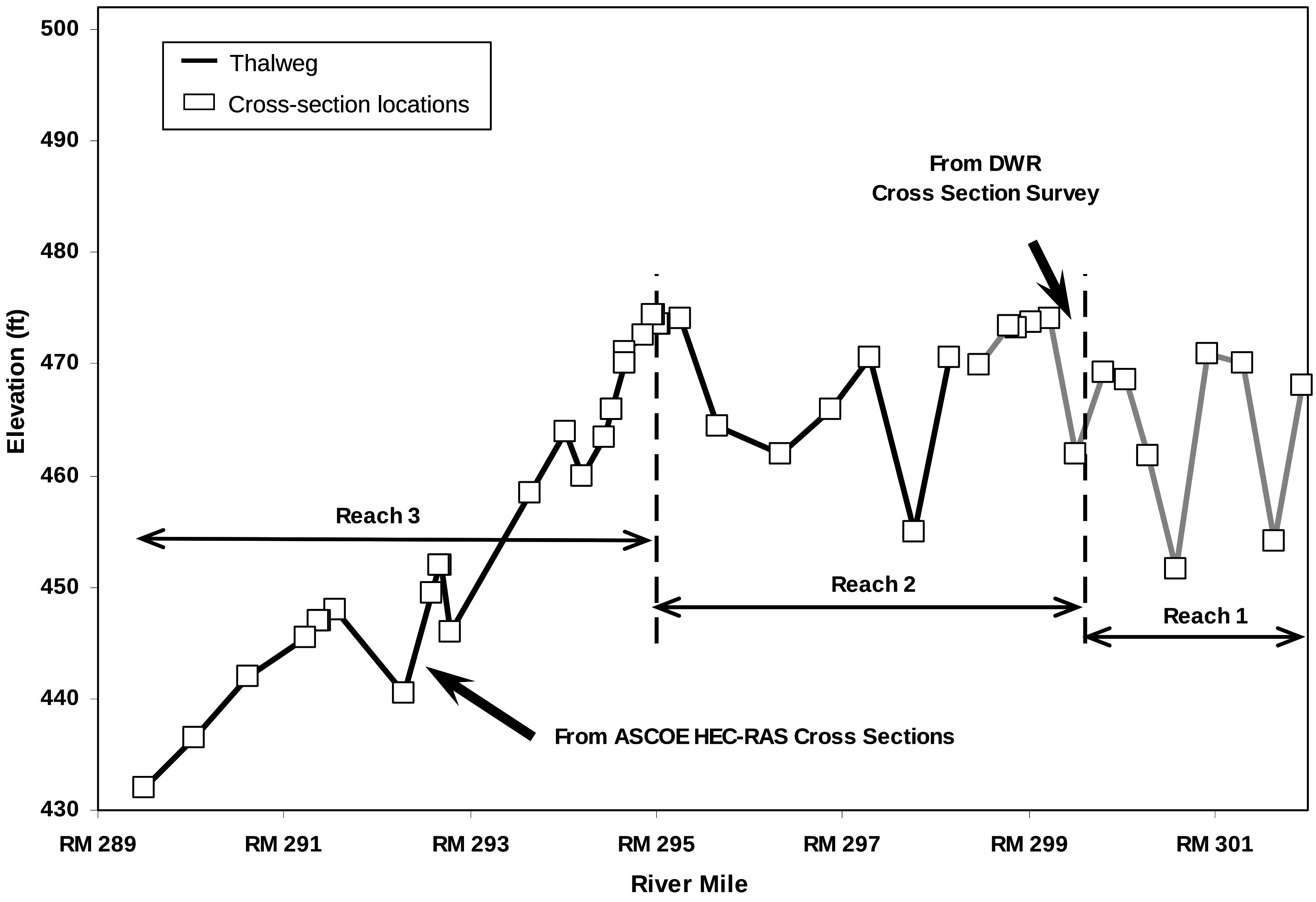


Figure 4.2-19. Thalweg profile in three study reaches between RM 289 and 301.

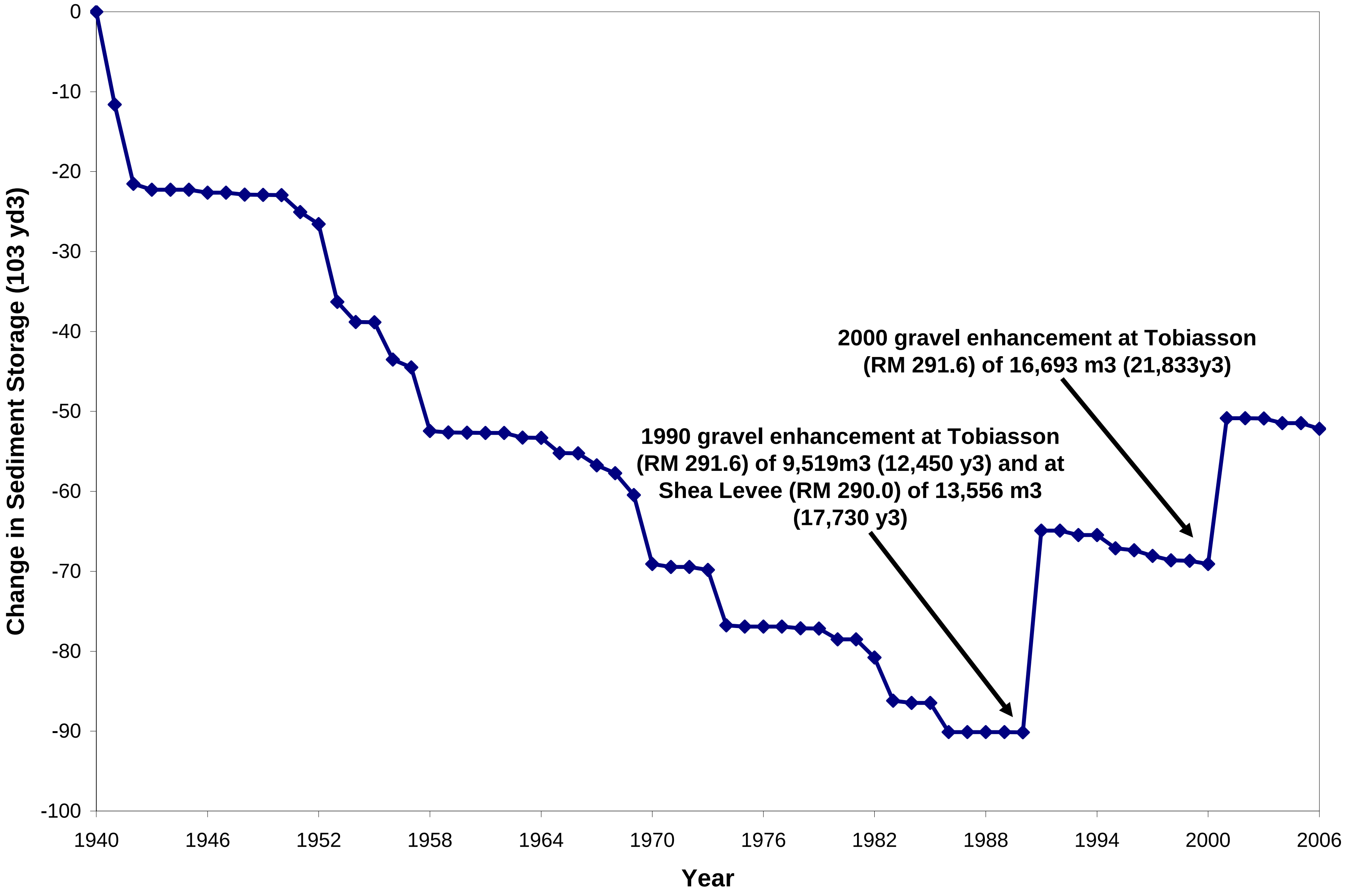


Figure 4.2-20. Simulated change in sediment storage in Subreach 3, Sacramento River (RM 290−295).

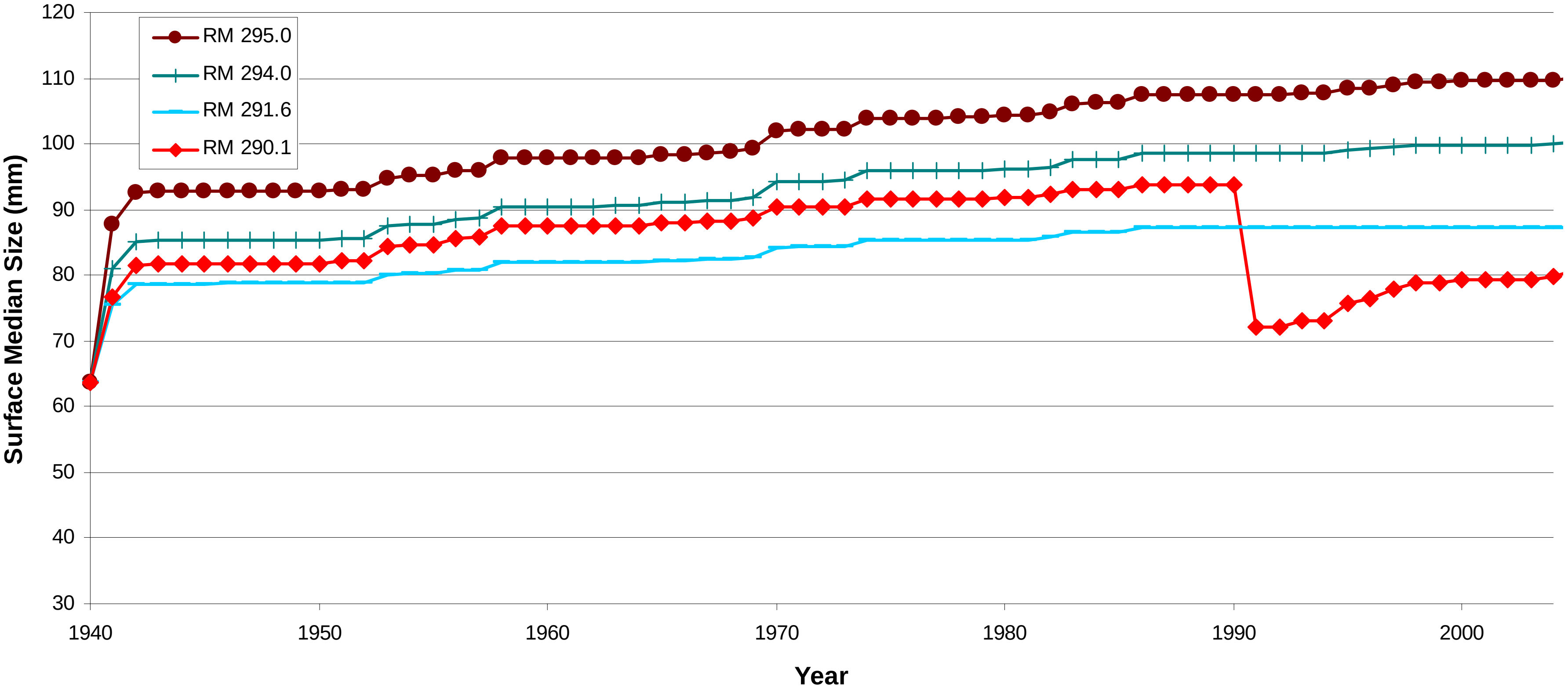


Figure 4.2-21. Simulated surface median size at four locations on the Sacramento River, beginning with a D_{50} of 63.7 mm (2.51 in).

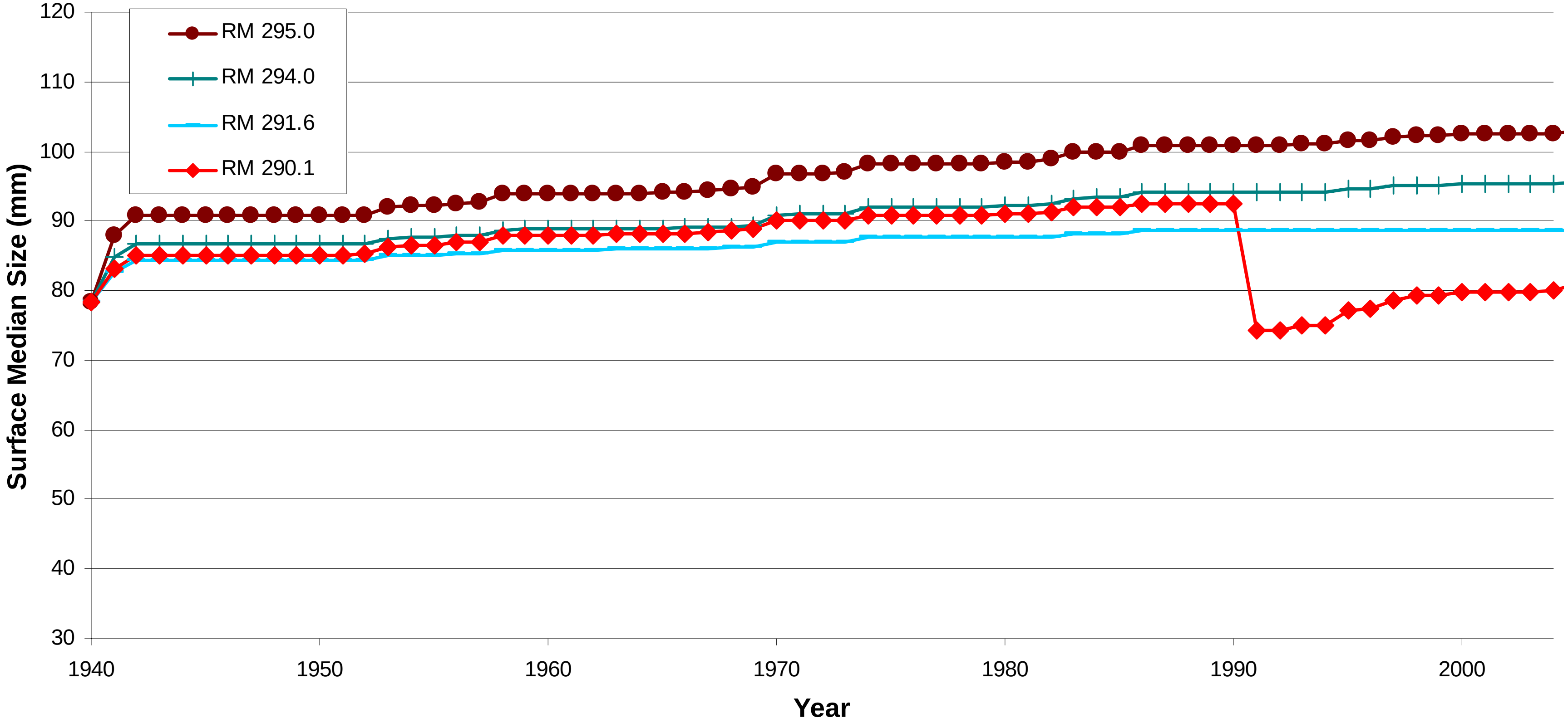


Figure 4.2-22. Simulated surface median size at four locations on the Sacramento River, beginning with a D50 of 78.4 mm (3.09 in). Using the TUGS model, we conducted a sensitivity analysis to assess the effects of a different initial grain size on bed coarsening. The results suggest that the grain size to which the bed will eventually coarsen is relatively insensitive to differences in the initial grain size.

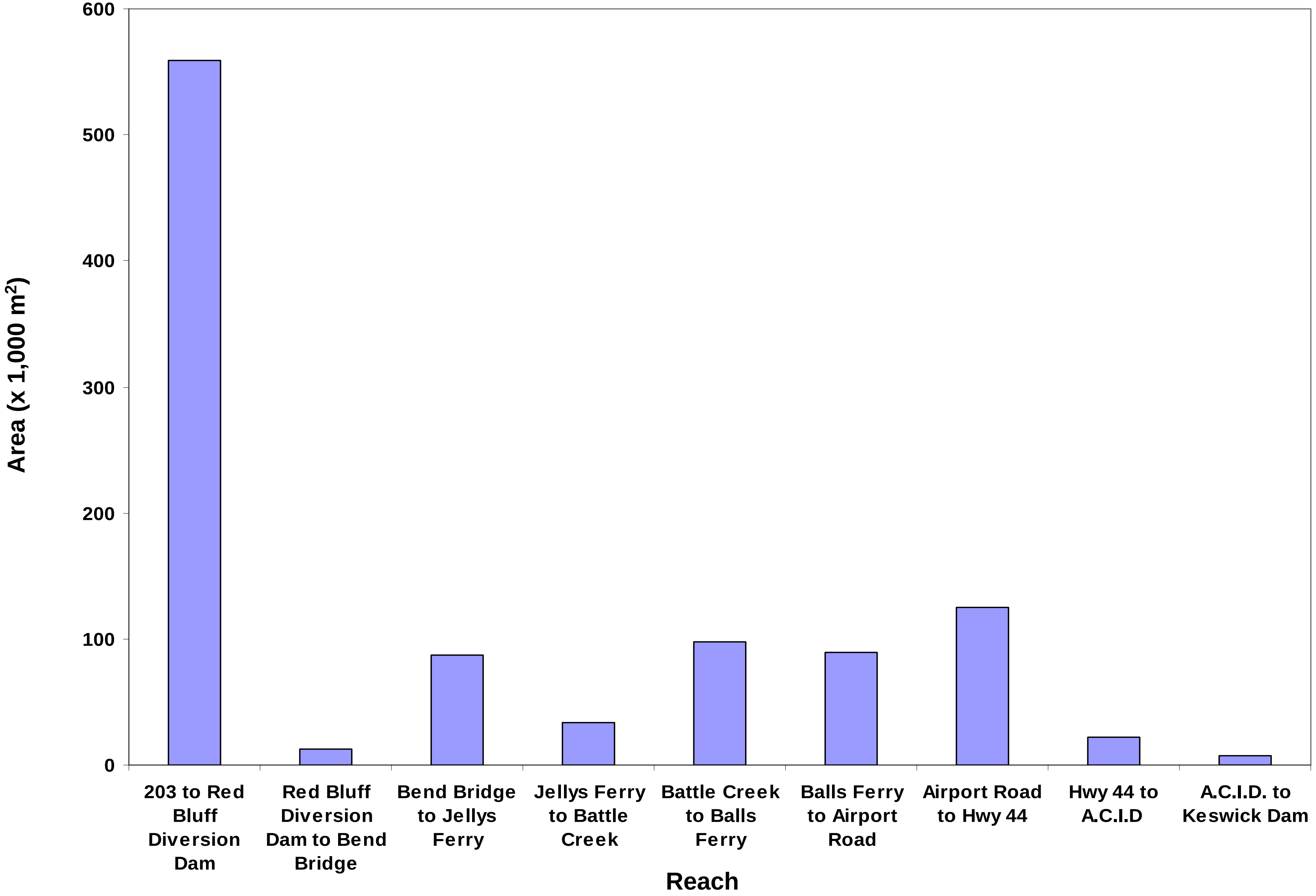


Figure 4.2-23. Spawning gravel distribution in the middle and upper Sacramento River as mapped in 1980.

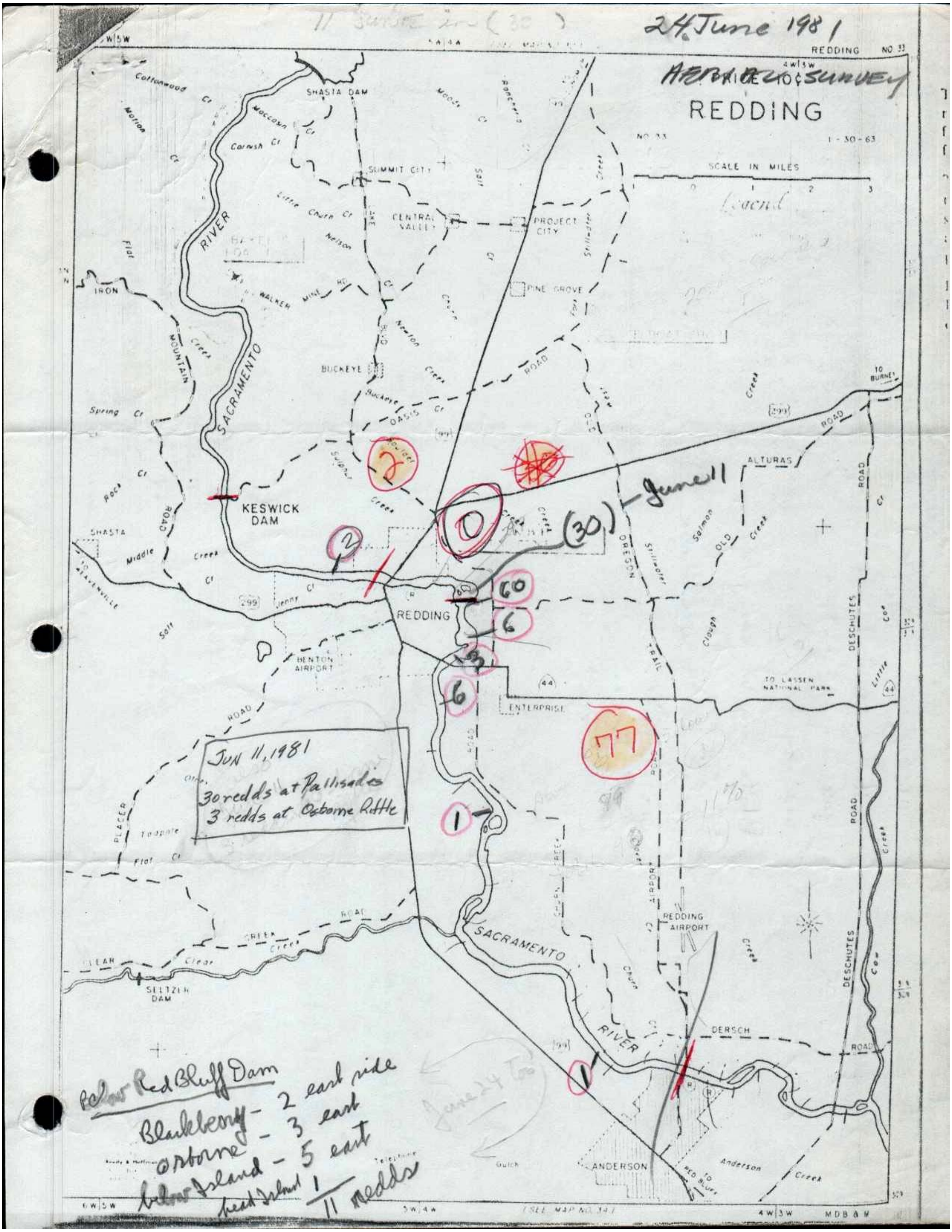


Figure 4.2-24. Aerial redd surveys below Keswick Dam. Source: CDFG 1981, unpublished data.

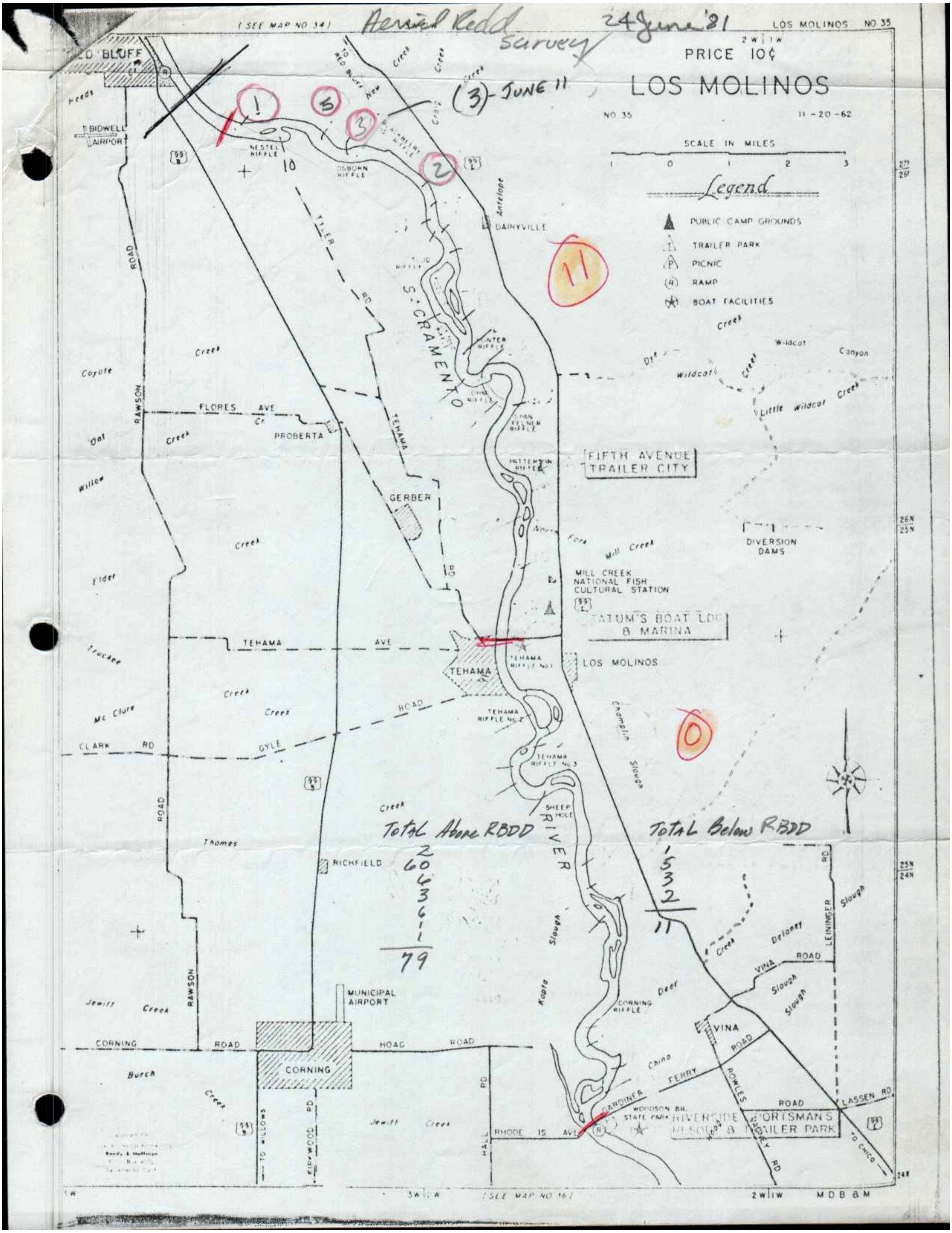


Figure 4.2-25. Aerial redd surveys below Red Bluff Diversion Dam. Source: CDFG 1981, unpublished data.

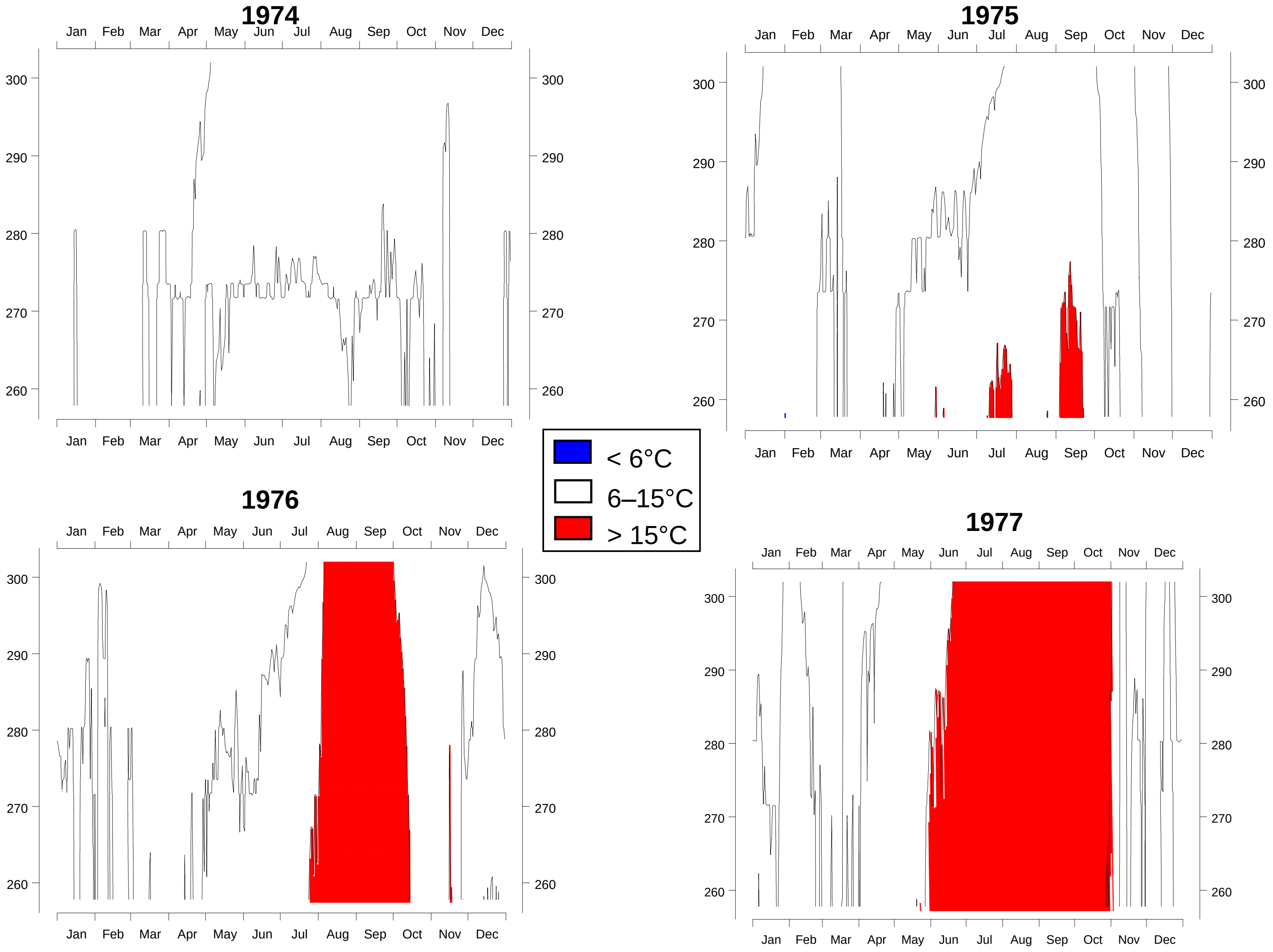


Figure 4.2-26. Temperatures on the Sacrament River between 1974 and 1977. Source: Watercourse Engineering 2002.

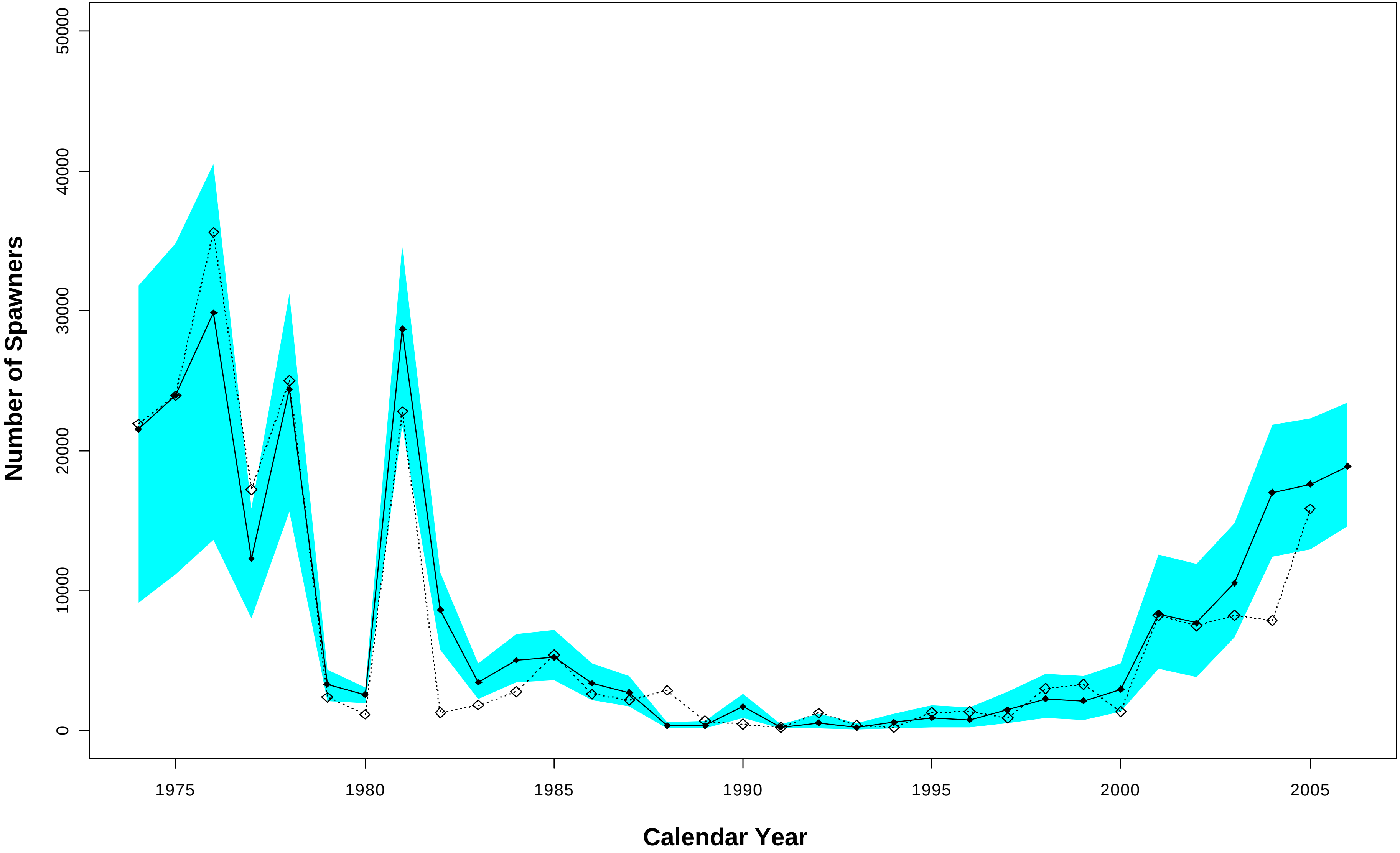


Figure 4.2-27. State-space model results with open diamonds representing historical escapements and solid diamonds representing model escapements. Blue region indicates the 95% confidence band for the predicted escapements.

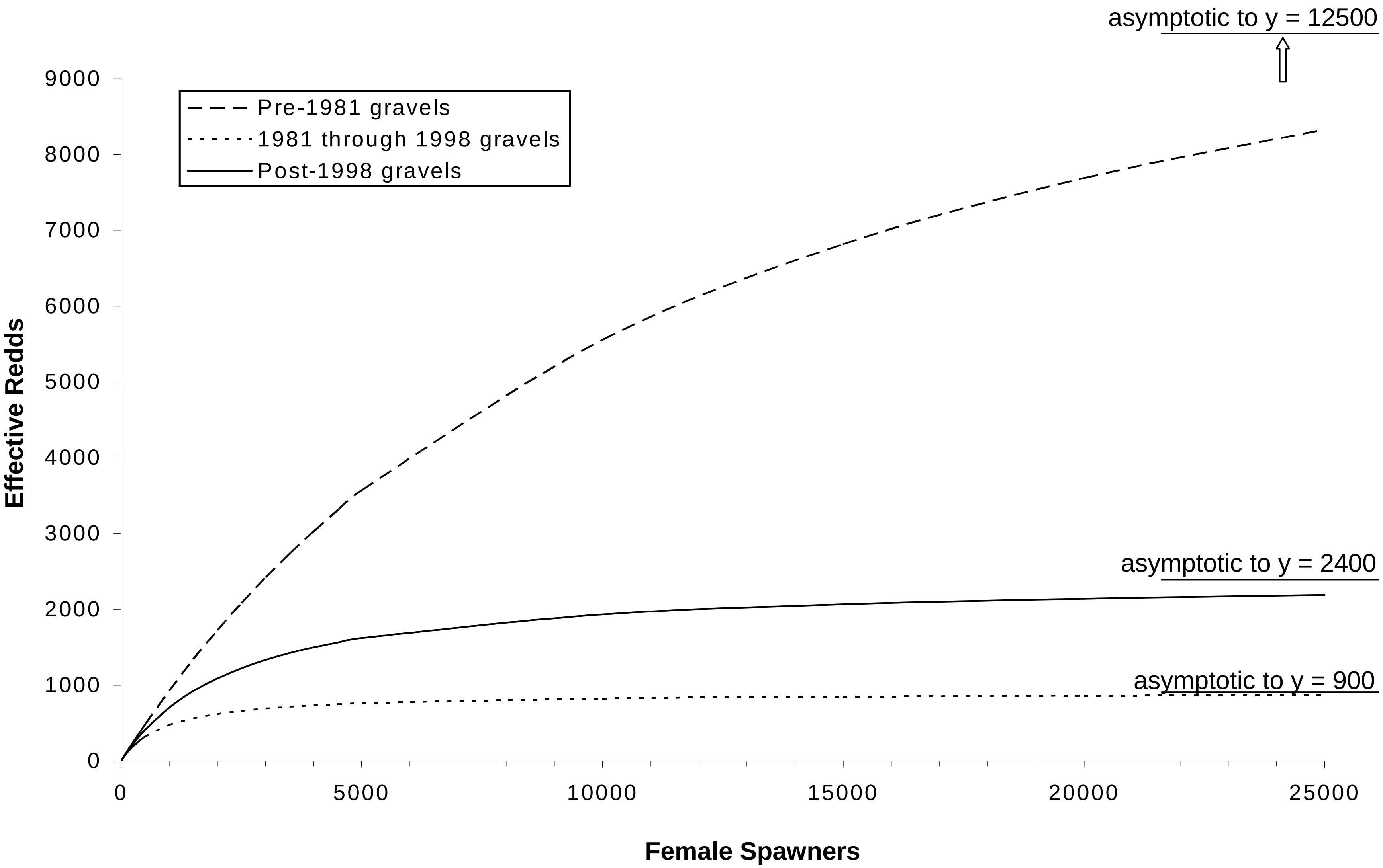


Figure 4.2-28. Number of effective redds, relative to the number of spawners. Effective redds are redds that are built and not superimposed.

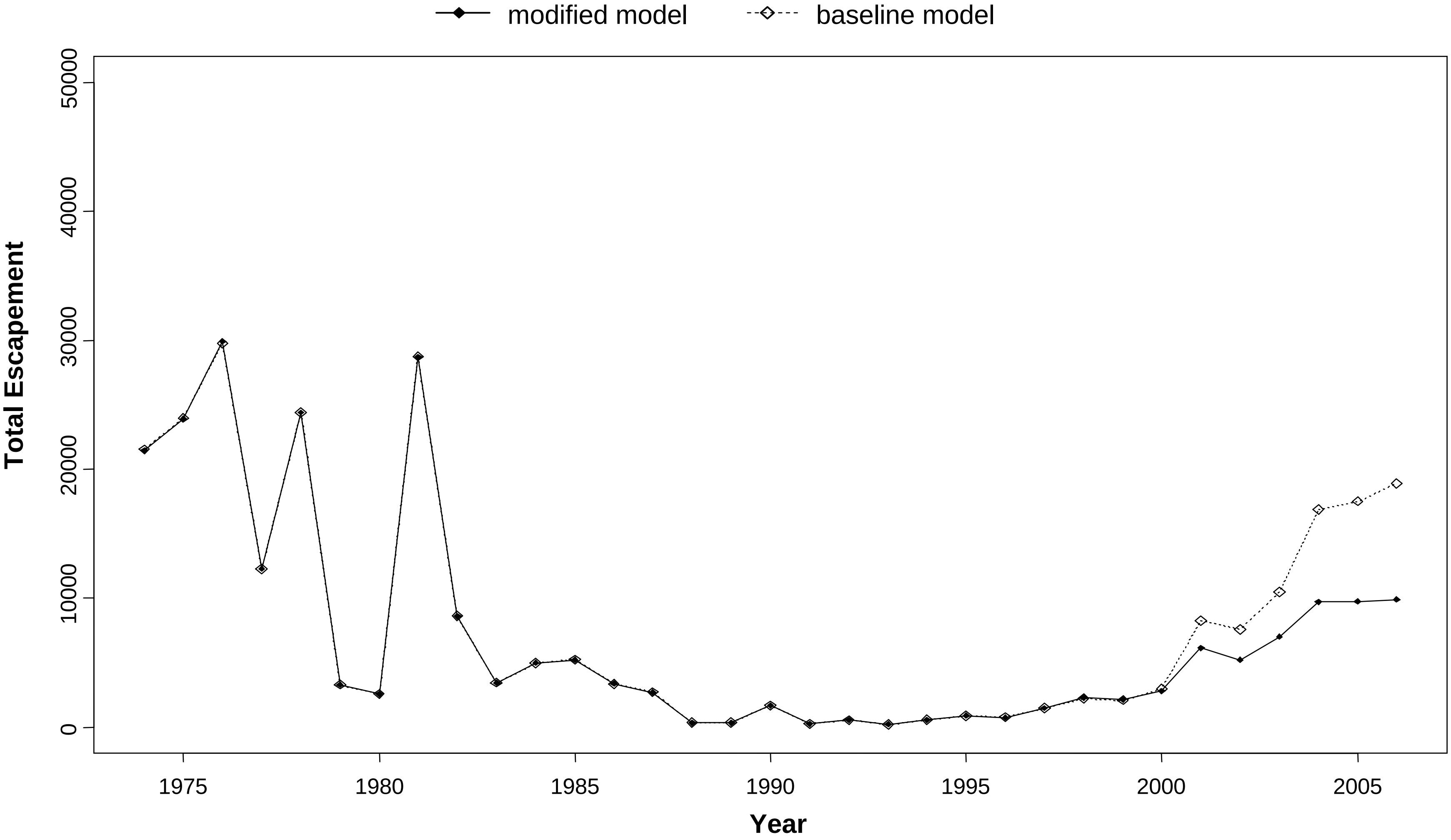
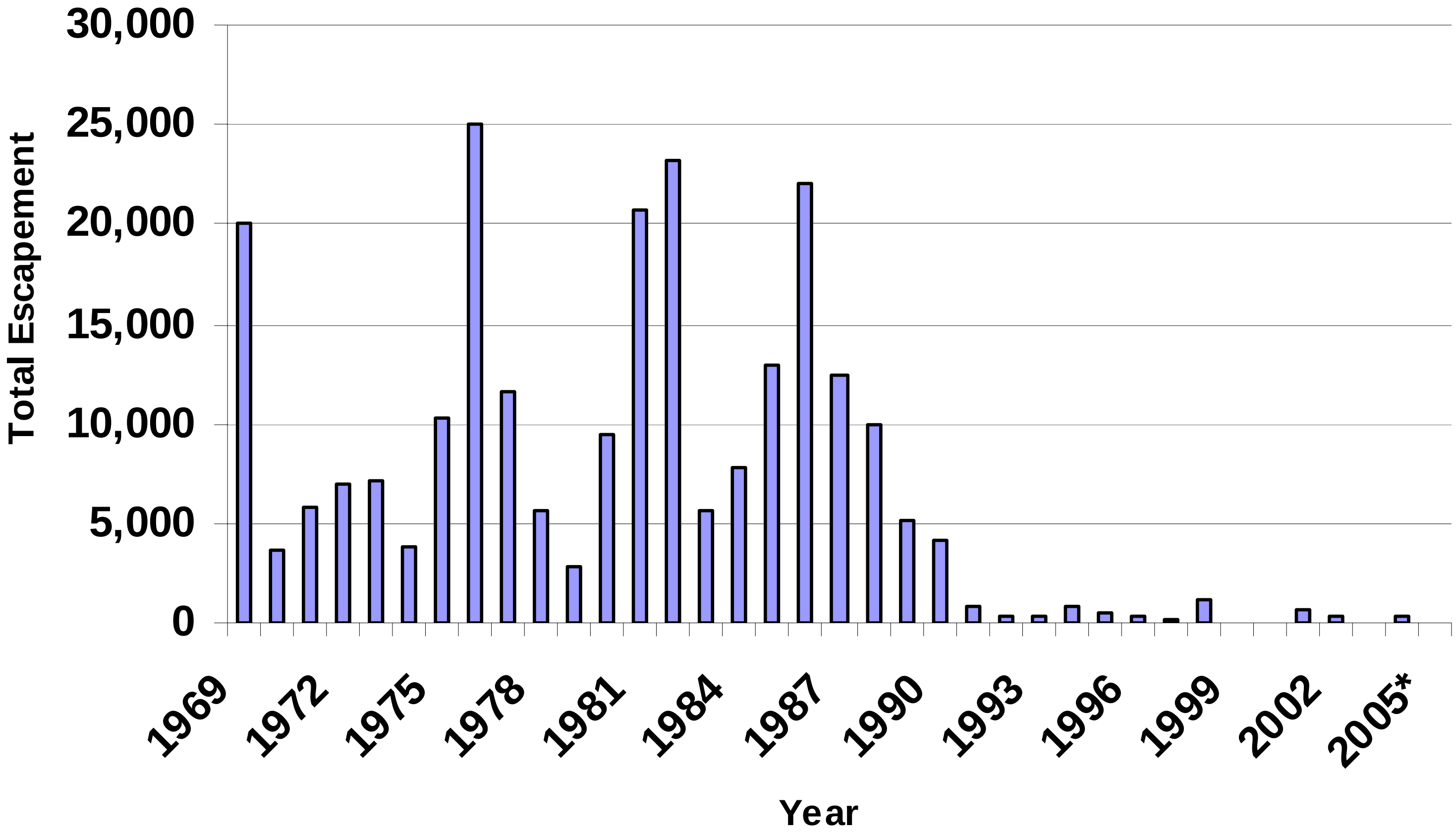


Figure 4.2-29. State-space model results showing the predicted benefits of gravel augmentation. Open diamonds represent historical escapement (1974 to present) and solid diamonds represent model results without gravel supplementation (1998 to present).



Mainstem Sacramento River	
Year	**Escapement**
1969	20,000
1970	3,652
1971	5,830
1972	7,038
1973	7,175
1974	3,800
1975	10,234
1976	25,095
1977	11,545
1978	5,669
1979	2,856
1980	9,369
1981	20,655
1982	23,156
1983	5,647
1984	7,823
1985	12,913
1986	22,058
1987	12,371
1988	9,867
1989	5,131
1990	4,198
1991	825
1992	371
1993	391
1994	862
1995	426
1996	378
1997	128
1998	1,115
1999	-
2000	71
2001	736
2002	273
2003	0
2004*	395
2005*	30

Figure 4.3-1. Spring-run Chinook salmon escapements between 1969 and 2005. The number of spring-run Chinook salmon that spawn in the mainstem Sacramento River has declined significantly since the mid-1980s, including years when no adults have been observed spawning in the mainstem channel. Source: GrandTab.xls.

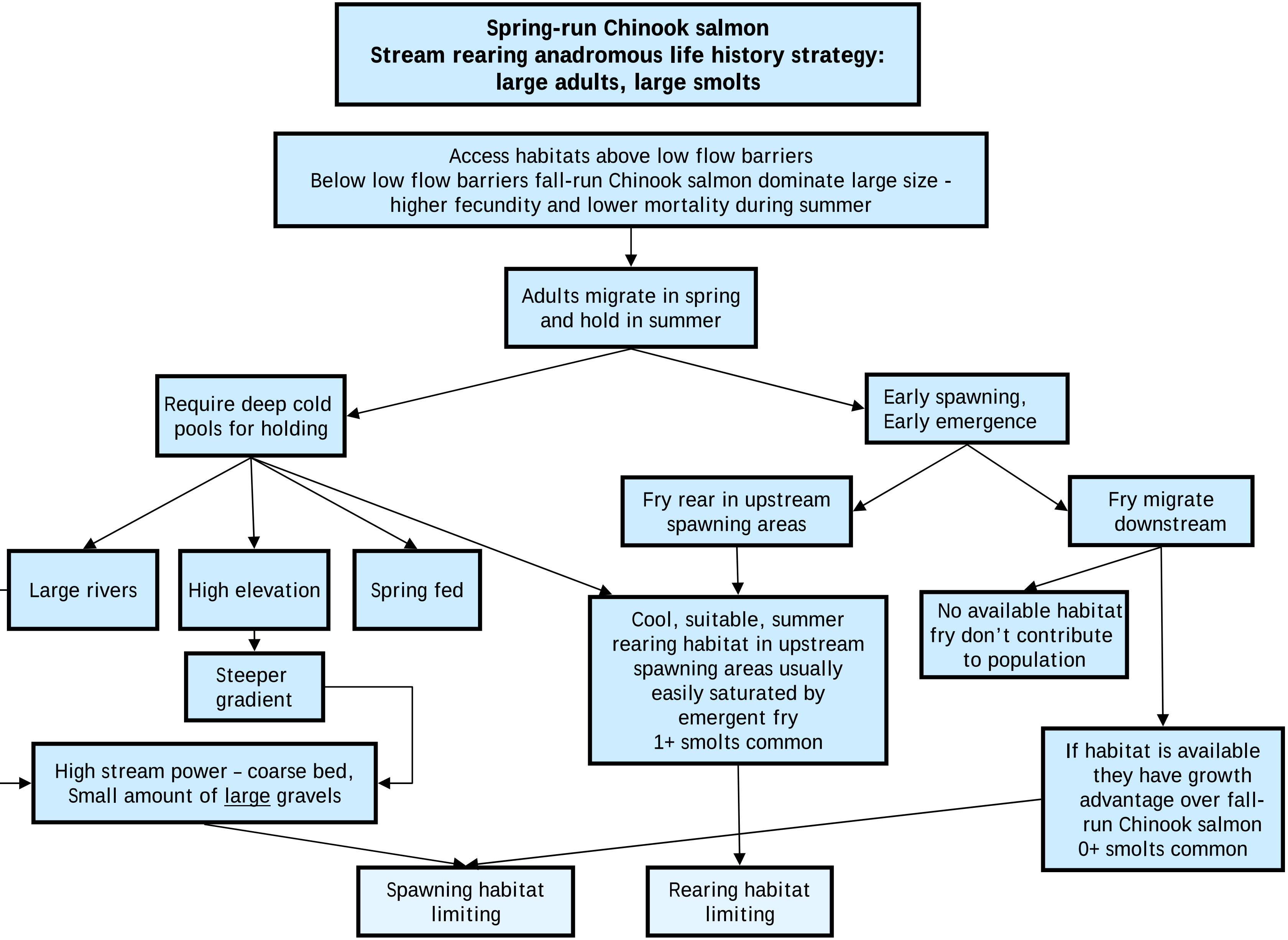
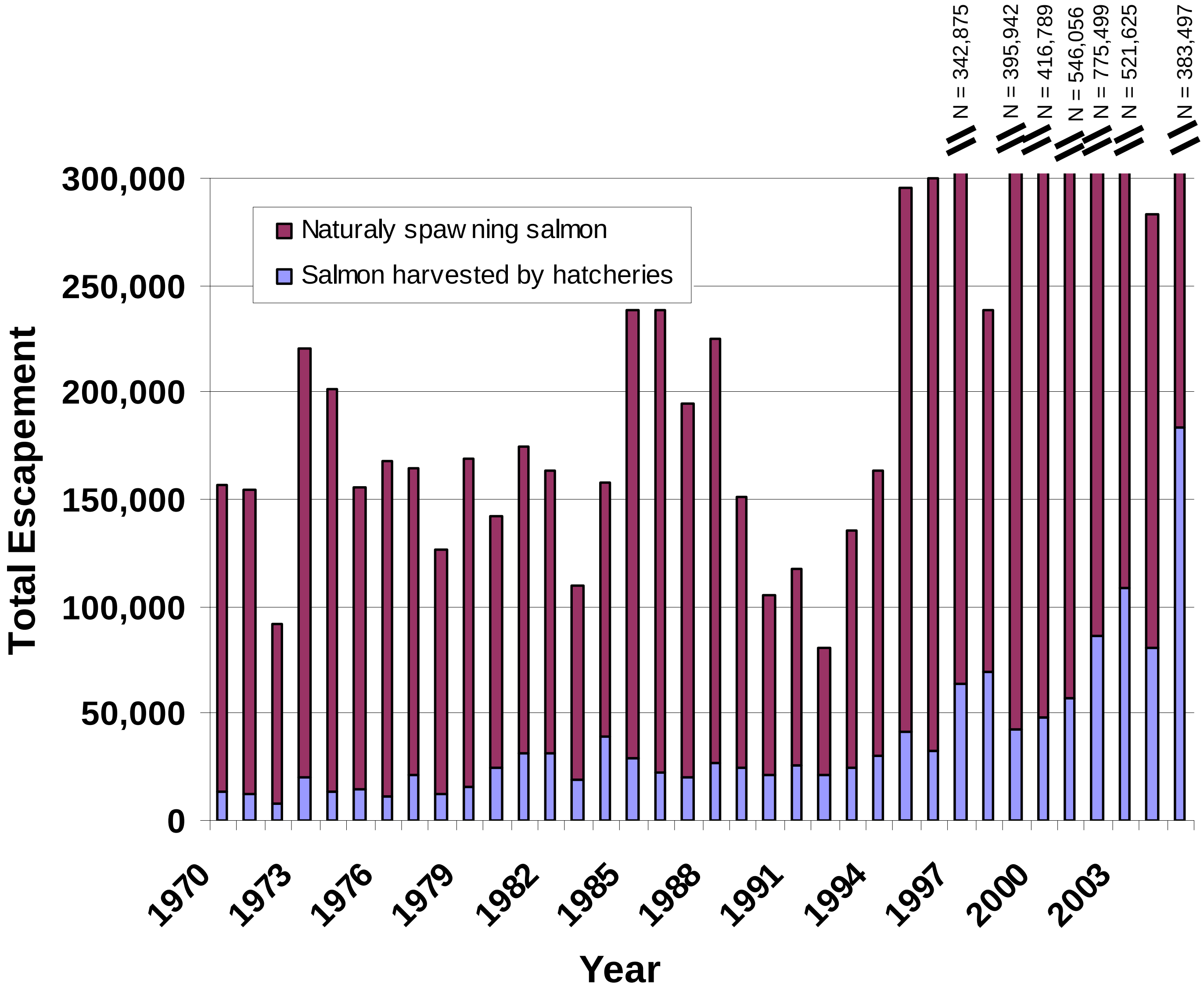


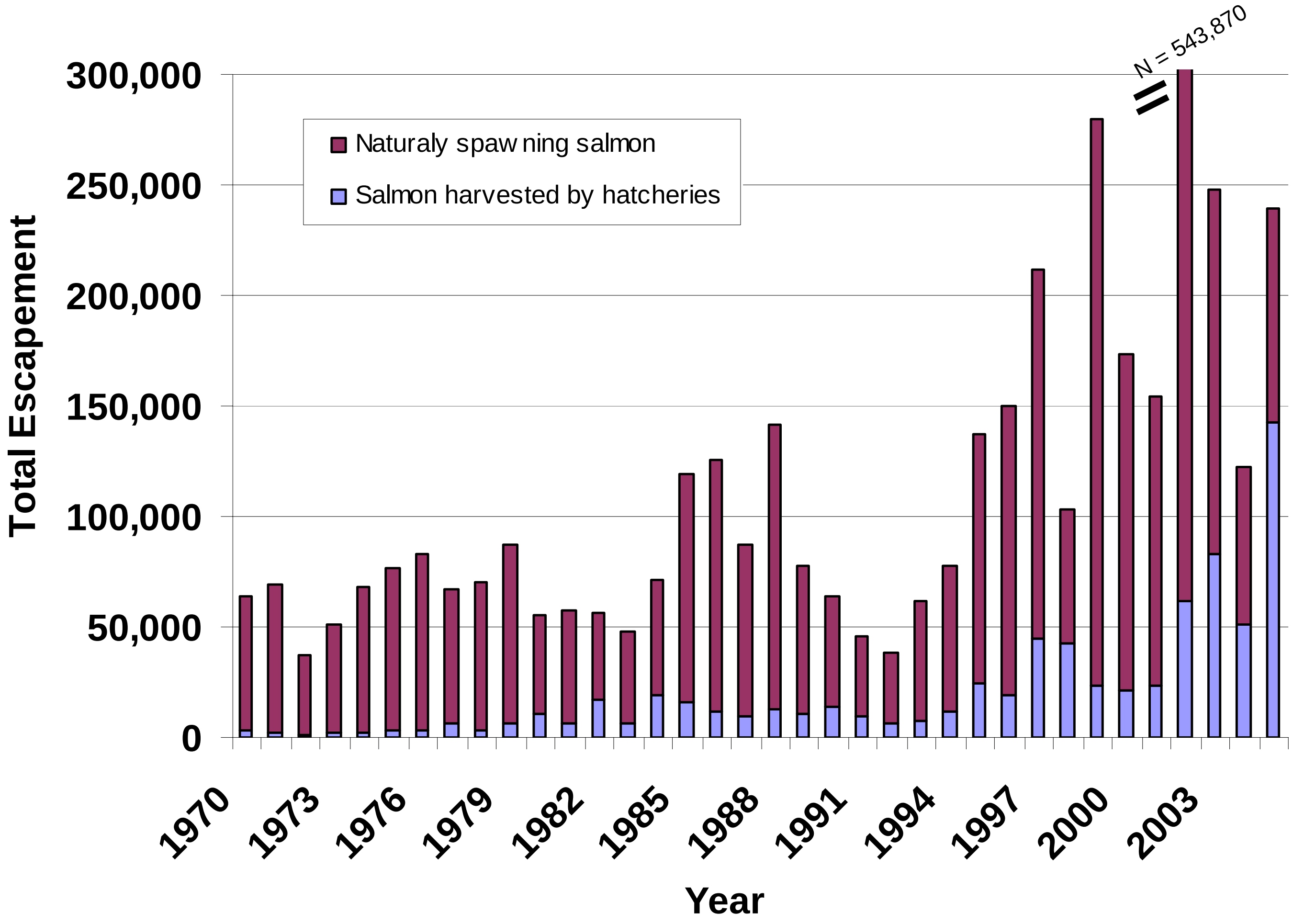
Figure 4.3-2. Spring-run Chinook salmon conceptual model of the Sacramento River.



Upper and lower Sacramento River			
Year	**Hatchery harvest**	**Natural spawning**	**Total**
1970	13,275	143,877	157,152
1971	12,739	142,143	154,882
1972	8,025	84,131	92,156
1973	19,689	200,371	220,060
1974	13,783	188,234	202,017
1975	14,071	141,550	155,621
1976	11,628	156,237	167,865
1977	20,978	143,032	164,010
1978	12,654	114,295	126,948
1979	15,812	153,632	169,444
1980	24,916	117,113	142,028
1981	30,930	143,961	174,891
1982	31,666	132,293	163,959
1983	18,586	90,800	109,386
1984	38,725	119,505	158,230
1985	29,254	209,450	238,704
1986	21,847	216,310	238,157
1987	19,833	174,791	194,623
1988	26,771	197,953	224,724
1989	24,942	126,683	151,625
1990	21,747	83,199	104,946
1991	26,030	91,402	117,432
1992	21,688	59,457	81,145
1993	24,626	110,556	135,182
1994	30,601	133,030	163,631
1995	41,548	253,486	295,034
1996	32,519	267,071	299,589
1997	63,276	279,600	342,875
1998	69,915	168,144	238,060
1999	42,224	353,718	395,942
2000	47,575	369,214	416,789
2001	57,399	488,657	546,056
2002	85,693	689,806	775,499
2003	108,198	413,427	521,625
2004	80,067	203,487	283,554
2005*	183,118	200,379	383,497

*Preliminary data

Figure 4.4-1. Annual escapements of fall-run Chinook salmon in the upper and lower Sacramento River basin (1970-2005). Escapement includes estimates of natural spawning salmon and salmon harvested by hatcheries for the fall-run Chinook salmon propagation program. Natural populations estimates were based on carcass surveys. Estimates between 1971-1985 included the Tehama-Colusa Spawning Channel. Source: PFMC 2006.



Upper Sacramento River			
Year	**Hatchery harvest**	**Natural spawning**	**Total**
1970	3,010	61,159	64,168
1971	1,728	67,586	69,314
1972	1,259	36,485	37,744
1973	1,679	48,948	50,627
1974	1,984	66,304	68,288
1975	3,289	72,996	76,275
1976	3,017	80,262	83,279
1977	6,083	60,966	67,049
1978	2,717	66,991	69,708
1979	6,407	81,332	87,739
1980	10,271	45,504	55,775
1981	5,883	51,832	57,714
1982	17,117	39,694	56,811
1983	6,112	41,969	48,082
1984	19,594	51,771	71,365
1985	15,869	103,698	119,566
1986	11,283	113,875	125,158
1987	9,981	76,861	86,824
1988	12,594	128,725	141,319
1989	10,212	67,296	77,508
1990	13,464	50,266	63,690
1991	10,031	35,258	45,289
1992	6,257	31,734	37,990
1993	7,056	55,144	62,200
1994	11,585	66,383	77,967
1995	24,810	112,234	137,044
1996	18,848	131,267	150,116
1997	44,590	167,354	211,943
1998	42,400	60,713	103,112
1999	23,194	256,629	279,823
2000	20,793	152,923	173,716
2001	23,710	130,440	154,150
2002	61,946	481,924	543,870
2003	82,708	164,802	247,510
2004	51,557	70,557	122,114
2005*	142,135	96,716	238,851

*Preliminary data

Figure 4.4-2. Annual escapements of fall-run Chinook salmon in the upper Sacramento River basin (1970-2005). Escapement includes estimates of natural spawning salmon and salmon harvested by hatcheries for the fall-run Chinook salmon propagation program. Natural populations estimates were based on carcass surveys. Estimates between 1971-1985 included the Tehama-Colusa Spawning Channel. Source: PFMC 2006.

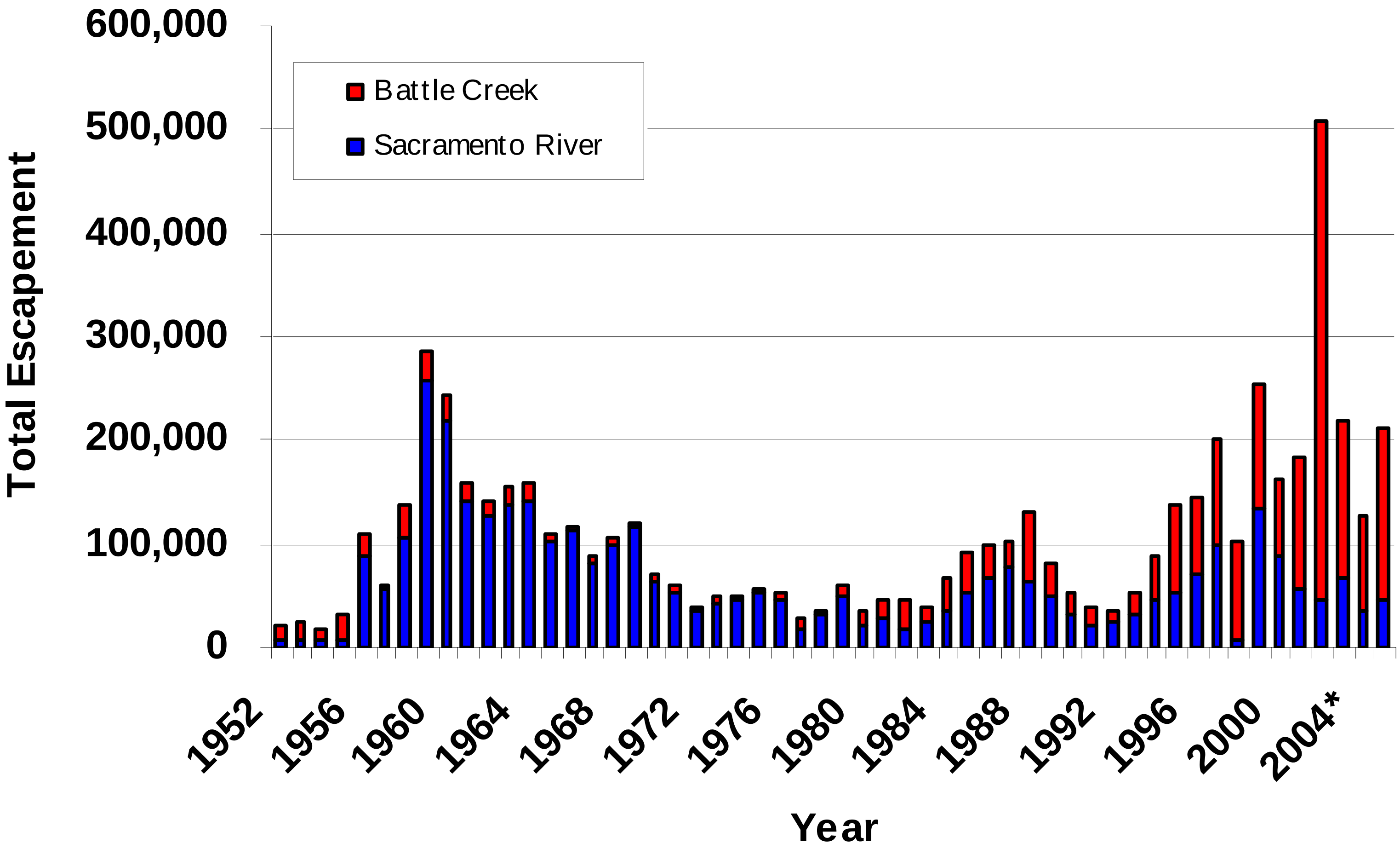


Figure 4.4-3. Annual escapements of fall-run Chinook salmon in the Sacramento River and Battle Creek (1952-2005). Source: GrandTab.xls.

Year	Sacramento River	Battle Creek
1952	7,000	15,000
1953	8,000	16,000
1954	6,000	12,000
1955	6,000	26,000
1956	87,357	21,108
1957	54,989	5,330
1958	107,153	29,243
1959	256,700	30,233
1960	218,940	23,805
1961	140,181	19,856
1962	127,837	13,057
1963	138,881	17,514
1964	142,584	15,875
1965	101,876	9,194
1966	111,881	3,300
1967	82,490	5,210
1968	98,429	6,476
1969	115,652	5,826
1970	65,142	6,832
1971	53,888	5,289
1972	33,958	4,852
1973	41,129	8,135
1974	47,019	3,901
1975	53,129	4,857
1976	45,753	5,444
1977	16,176	10,848
1978	32,235	3,652
1979	47,758	13,159
1980	21,961	14,443
1981	29,212	17,205
1982	17,966	26,795
1983	26,226	13,983
1984	36,965	29,893
1985	52,120	39,808
1986	68,821	31,252
1987	76,562	24,249
1988	63,998	67,475
1989	48,968	31,048
1990	32,109	21,088
1991	20,523	17,241
1992	23,914	12,708
1993	33,471	18,616
1994	44,729	43,265
1995	53,385	83,192
1996	71,725	73,587
1997	98,765	101,414
1998	5,718	98,308
1999	133,365	119,899
2000	87,793	75,106
2001	57,792	125,686
2002	45,523	463,296
2003	66,476	153,045
2004*	34,050	92,090
2005*	44,950	165,259

* Preliminary data

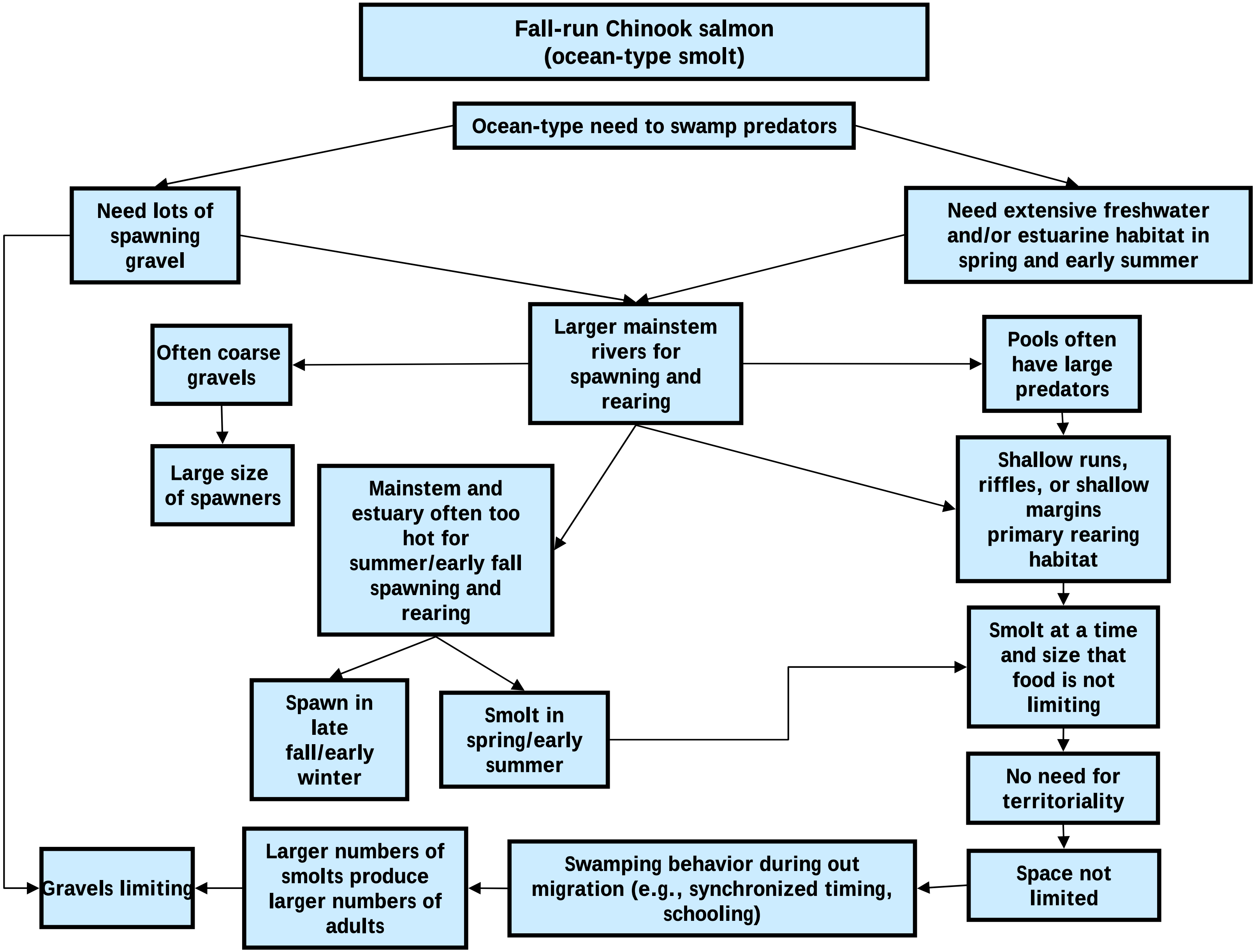
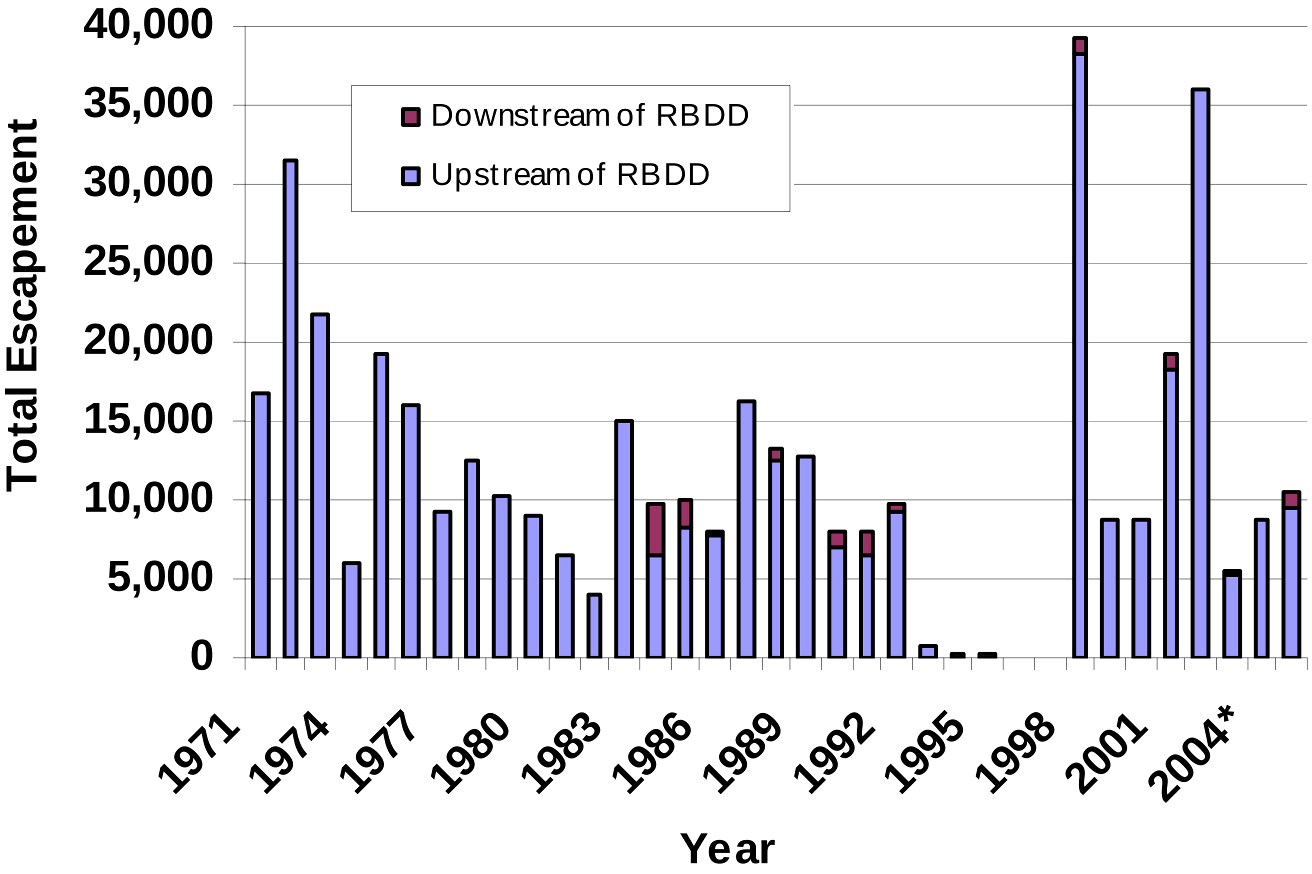


Figure 4.4-4. Fall-run Chinook salmon conceptual model of the Sacramento River.



Year	Upstream of RBDD	Downstream of RBDD
1971	16,741	-
1972	31,559	-
1973	21,781	-
1974	6,083	-
1975	19,261	-
1976	15,908	-
1977	9,210	-
1978	12,479	-
1979	10,284	-
1980	9,093	-
1981	6,571	-
1982	3,981	-
1983	14,984	-
1984	6,540	3,098
1985	8,136	1,863
1986	7,820	284
1987	16,222	-
1988	12,507	658
1989	12,807	0
1990	6,892	1,094
1991	6,611	1,491
1992	9,356	431
1993	739	-
1994	291	-
1995	166	-
1996	48	-
1997	-	-
1998	38,239	1,101
1999	8,683	-
2000	8,632	119
2001	18,351	925
2002	36,004	0
2003	5,346	148
2004*	8,824	0
2005*	9,565	1,035

*Preliminary data

Figure 4.5-1. Annual escapements of late-fall-run Chinook salmon in the Sacramento River above and below the Red Bluff Diversion Dam (1971-2005). Source: GrandTab.xls.

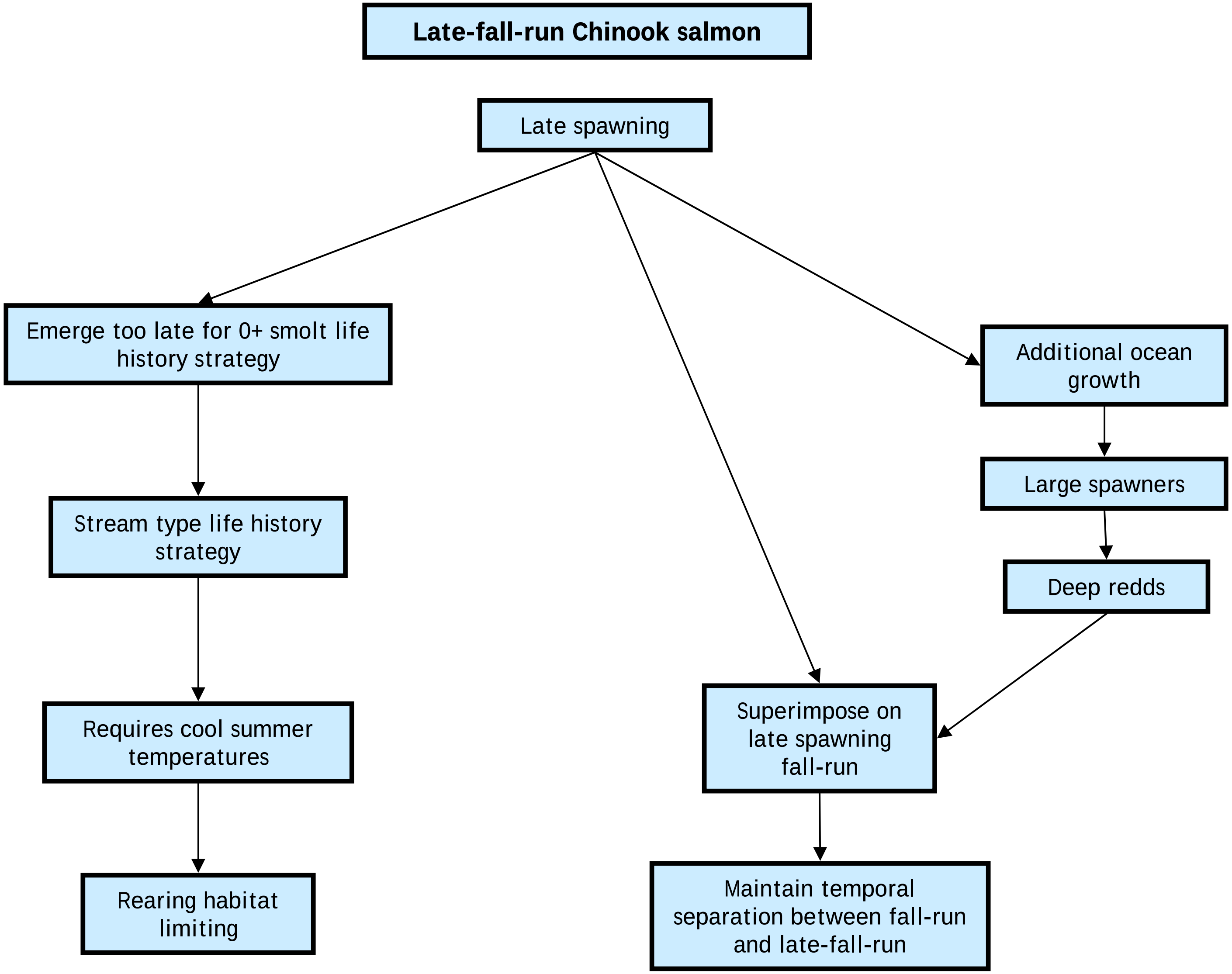
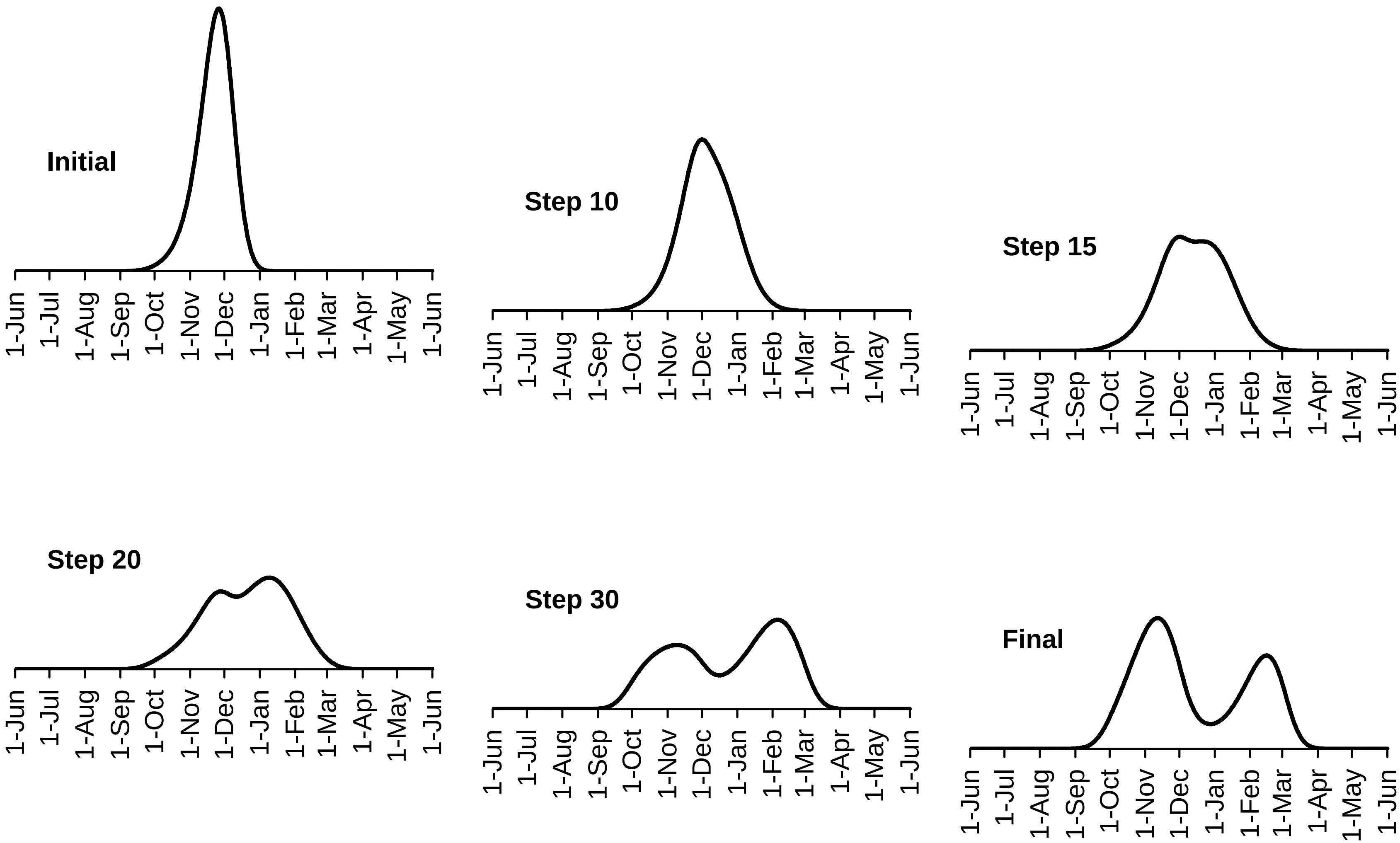


Figure 4.5-2. Late-fall-run Chinook salmon conceptual model of the Sacramento River.



Initial

Step 10

Step 15

Step 20

Step 30

Final

Figure 4.5-3. Splitting a stable fall-run Chinook salmon pattern into stable fall and late-fall run patterns after enabling a new life history strategy, juvenile over-summering.

5 CALIFORNIA CENTRAL VALLEY STEELHEAD *(ONCORHYNCHUS MYKISS)*

California Central Valley Steelhead pose a difficult management challenge in the Sacramento River. There has been only limited research and monitoring in comparison with Chinook salmon, so there is little specific information about the status and trend of the species and how adults and juveniles use habitats in the mainstem river and the Bay-Delta estuary. Though the upper reaches of the Sacramento River support a spawning population of resident rainbow trout, the mainstem river habitat used by the species is atypical for steelhead, which usually spawn in higher elevation, steeper, and narrower channels. Management of the species is also complicated by its polymorphism, with individuals being capable of exhibiting either a resident (e.g., rainbow trout) or an anadromous (e.g., steelhead) life history.

NOAA Fisheries listed the California Central Valley Steelhead Distinct Population Segment (DPS) as threatened under the Federal Endangered Species Act in 1998 (NMFS 1998).

5.1 Distribution

5.1.1 Historical distribution in the Central Valley

O. mykiss once occurred throughout the Central Valley, spawning in the upper reaches of tributaries to the Sacramento and San Joaquin rivers. Lindley et al. (2006) recently conducted GIS-based habitat modeling to estimate the amount of suitable habitat to support *O. mykiss* populations in the Central Valley, and their results suggest that steelhead were widely distributed throughout the Sacramento River basin, but relatively less abundant in the San Joaquin River basin due to natural barriers to migration. Yoshiyama et al. (1996) conducted a thorough review of historical sources to document the historical distribution of Chinook salmon in the Central Valley, which can be used to infer historical distribution of steelhead. The assumption that steelhead distribution in the Sacramento River basin overlapped with, and was likely more extensive than, spring-run Chinook distribution under historical conditions has been supported by studies conducted in the Klamath-Trinity river basin (CH2M Hill 1985, Voight and Gale 1998). Yoshiyama et al. (1996) concluded that, because steelhead upstream migration occurs during high flows, their leaping abilities are superior to those of Chinook salmon, and they have less restrictive spawning gravel criteria, steelhead in the Sacramento River basin "could have used at least hundreds of miles of smaller tributaries not accessible to the earlier-spawning salmon." The model created by Lindley et al. (2006) estimates that 80% of historically accessible habitat for Central Valley steelhead is now behind impassable dams; this figure is supported by other research into steelhead and Chinook salmon habitat loss in the Central Valley (Clark 1929, Yoshiyama et al. 1996, 2001).

5.1.2 Current distribution in the Sacramento River basin

In the Sacramento River basin, populations of *O. mykiss* are known to spawn in the upper Sacramento, Yuba, Feather, and American rivers, and in Deer, Mill, and Butte creeks. Saeltzer Dam was removed from Clear Creek in 2000, granting easier access to upstream habitats in the canyon reaches of the creek. Though improved access may have opened up suitable spawning and rearing habitat for steelhead, it is not clear if steelhead have colonized Clear Creek since removal of the dam. A summary of recent distribution information for steelhead in Sacramento River tributaries is shown in Table 62 of Good et al. (2005), which shows that steelhead are widespread in accessible streams, if not abundant.

5.2 Population Trends

In general, steelhead stocks throughout California have declined substantially. McEwan and Jackson (1996) reported that the adult population of steelhead in California was roughly 250,000, less than half the population that existed in the 1960s (McEwan and Jackson 1996). In the Central Valley, roughly 1–2 million adult steelhead may have returned annually prior to 1850, as based on historical Chinook salmon abundance (McEwan 2001, NMFS 2006). In the Sacramento River basin, the average run size of steelhead in the 1950s was estimated to be approximately 20,540 adults (McEwan and Jackson 1996). In contrast, escapement estimates in 1991 and 1992 were less than 10,000 adults, or less than half of the run size in the 1950s (McEwan and Jackson 1996). Similarly, counts of wild steelhead at Red Bluff Diversion Dam (RBDD) declined from an average annual run size of 12,900 in the late 1960s to 1,100 adults in the 1993–1994 season (McEwan and Jackson 1996). The most recent 5-year average for steelhead spawning upstream of Red Bluff Diversion Dam is less than 2,000 adults (Good et al. 2005). NMFS (2006) notes that there have not been any escapement estimates made for the area upstream of RBDD since the mid-1990s, and that estimates of abundance are currently derived from extrapolation of incidental catch of outmigrating juvenile steelhead captured as part of the midwater-trawl sampling for juvenile Chinook salmon at Chipps Island, downstream of the confluence of the Sacramento and San Joaquin rivers.

5.3 Life History

There are generally two types of steelhead: winter steelhead and summer steelhead. Winter steelhead become sexually mature during their ocean phase and spawn soon after arriving at their spawning grounds. Adult summer steelhead enter their natal streams and spend several months holding and maturing in fresh water before spawning. California Central Valley steelhead are predominantly winter steelhead; consequently, this section describes the life history and habitat requirements of winter steelhead.

It is worth noting that summer steelhead occur in coastal tributaries of northern California, and some investigators hypothesize that summer steelhead may have been more prevalent in California before larger dams eliminated access to historical holding habitat (McEwan 2001). Like summer steelhead, spring-run Chinook salmon are also stream-maturing; their historical distribution throughout Central Valley tributaries suggests that there was habitat available to support the life history strategy of summer steelhead as well (e.g., deep, coldwater holding pools located in high elevation reaches). Although the availability of suitability habitat is not proof that summer steelhead were present, there appear to be at least a few records of summer steelhead from fish counts conducted in the Sacramento River system from before the large dams were constructed (Needham et al. 1941, USFWS and CDFG 1953; both as cited in McEwan 2001). NMFS (1998) notes that three distinct runs of steelhead may have been present in the Sacramento River basin as recently as 1947, including a summer run in the American River (Cramer et al. 1995, McEwan and Jackson 1996).

The management of steelhead populations in Central Valley tributaries is usually subsumed within the management of Chinook salmon populations because of their similar life history strategies and habitat requirements. Nevertheless, steelhead generally exhibit a more flexible life history strategy than Chinook salmon, and the habitat requirements of juvenile steelhead differ from those of juvenile Chinook.

Steelhead migrate up the Sacramento River nearly every month of the year, with the bulk of migration occurring from August through November, and the peak in late September (Bailey 1954; Hallock et al. 1961, both as cited in McEwan and Jackson 1996; McEwan 2001). Spawning in the upper Sacramento River generally occurs between November and late April, with a peak between early January and late March (USBR 2004). Fry emergence is influenced by water temperature, but hatching generally requires four weeks, with another four to six weeks in the gravels before emergence. Juvenile steelhead typically rear in freshwater from 1 to 3 years before emigrating (McEwan and Jackson 1996). The majority of returning adult steelhead in the Central Valley have spent two years in fresh water before emigrating to the ocean (McEwan 2001). A scale analysis conducted by Hallock et al. (1961, as cited in McEwan 2001) indicated that 70% emigrated after two years, 29% after one year, and 1% after three years in fresh water. Juvenile emigration from the upper Sacramento River occurs between November and late June, with a peak between early January and late March (USBR 2004).

Unlike Chinook salmon, steelhead can be iteroparous, which means that they can survive spawning, return to the ocean, and then migrate into fresh water to spawn again. Post-spawning adults are known as kelts; although some kelts have been documented in the Sacramento River, there are probably few repeat spawners in the Sacramento River population (USBR 2004).

In coastal populations of winter steelhead, it is a common life history strategy for juvenile steelhead to migrate downstream at age 1+ and rear in the estuary for an additional year before smolting. Some of the age 1+ steelhead captured in rotary screw traps t RBDD, GCID, and Knights Landing may continue rearing for another year before entering the ocean, but little information is available regarding steelhead use of the Sacramento-San Joaquin Delta estuary as rearing habitat. In addition, the potential effects of Delta water operations on steelhead have not been evaluated (McEwan 2001). There may be some areas of the Bay-Delta estuary where summer water temperatures are moderated by tidal action so that steelhead 1+ migrants are able to rear throughout the summer; however, this is currently an uncertainty that requires additional research.

5.4 Habitat Requirements

5.4.1 Spawning habitat

O. mykiss currently spawn in the mainstem Sacramento River below Keswick Dam (RM 302), with peak spawning occurring from January through March when water temperatures throughout much of the Sacramento River are suitable to support egg incubation and emergence. However, the downstream extent of spawning is likely determined by the location of suitable water temperatures to support summer rearing of 0+ juveniles, which lack the swimming ability to move significant distances upstream to follow the upstream retreat of cold water in the summer. The progeny of any adults that construct redds downstream of locations with suitable water temperatures in the summer likely suffer high rates of mortality and contribute little to the population.

Unlike the annual redd surveys conducted by CDFG to document the spawning locations of Chinook salmon, no regular surveys are conducted to document locations of *O. mykiss* spawning in the Sacramento River. Steelhead migrate and spawn during high flows when observations and sampling are difficult (McEwan 2001). It may be possible to use late-fall-run Chinook salmon spawning distribution as a proxy for steelhead spawning distribution, because the two species have similar juvenile life history strategies (juveniles rear in the river for at least one summer before emigrating), and redds must be located where summer water temperatures are suitable to support summer rearing. As discussed in Chapter 4.5, we hypothesize that the downstream extent of late-fall run Chinook spawning is generally located near Ball's Ferry Bridge (RM 276) in most years because this area defines the location of suitable summer water temperatures to support summer rearing. Steelhead generally have higher thermal tolerances than Chinook salmon (Moyle 2002), so the downstream extent of steelhead spawning may be slightly further downstream than for Chinook salmon.

As with Chinook salmon, steelhead spawn in areas with suitable gravel and hydraulics. Bovee (1978) reports that steelhead prefer water depths of 14 in (36 cm) for spawning, with a range between 6 and 24 in (15 and 61 cm), and water velocities of 2 ft/sec (61 cm/s), with a range of 1 to 3.6 ft/sec (30 to 110 cm/s), which is similar to the hydraulic conditions preferred by Chinook salmon in the Central Valley. As with Chinook salmon, steelhead generally prefer to spawn in gravels, with optimal grain sizes reported to range between 0.6 cm and 10 cm (6 mm and 102 mm) (Bjornn and Reiser 1991). Grain sizes used by spawning Chinook have been found to range from a D_{50} of 0.43 in (10.8 mm) (Platts et al. 1979, as cited in Kondolf and Wolman 1993) to a D_{50} of 3.1 in (78.0 mm) (Chambers et al. 1954, 1955, as cited in Kondolf and Wolman 1993).

Under historical conditions, steelhead likely spawned in much higher-gradient reaches in the Sacramento River and its tributaries, as do steelhead in other portions of their range. Steelhead are common in reaches with gradients of less than 6% (Burnett 2001, Harvey et al. 2002, Hicks and Hall 2003; all as cited in Lindley et al. 2006), and occur in some systems in reaches of up to 12% and more (Engle 2002, as cited in Lindley et al. 2006).

There is no Sacramento-specific information about water temperature requirements for successful spawning and incubation, but values derived from other steelhead stocks in more northerly locations suggest optimal spawning temperatures are between 39°F (4°C) and 52°F (11°C), with egg mortality occurring at water temperatures above 56°F (13°C) (Hooper 1973, Bovee 1978; Reiser and Bjornn 1979; Bell 1986; all as cited in McEwan and Jackson 1996). More research is needed to understand the specific temperature tolerances of steelhead in the Central Valley and southern portions of their range. There is some evidence that different strains of *O. mykiss* may have different thermal tolerances at the egg and embryo stage (Myrick and Cech 2001).

5.4.2 Summer rearing habitat

After emerging, steelhead fry typically disperse to shallow (< 14 in [36 cm], low-velocity near-shore areas such as stream margins and low-gradient riffles and will forage in open areas lacking instream cover (Hartman 1965, Everest et al. 1986, Fontaine 1988). Everest and Chapman (1972) found that juvenile steelhead of all sizes most often chose territories over large-sized substrates. As they increase in size in the late summer and fall, they increasingly use areas with cover and show a preference for higher-velocity, deeper mid-channel areas near the thalweg (Hartman 1965, Everest and Chapman 1972, Fontaine 1988). Bovee (1978) reports that fry prefer water depths of 10 in (25 cm), with a range between 10 in (25 cm) and 20 in (51 cm) and water temperatures ranging between 45°F (7°C) and 60°F (16°C). Age 0+ steelhead have been found to be relatively abundant in backwater pools and often live in the downstream ends of pools in late summer (Bisson et al. 1988, Fontaine 1988).

Steelhead fry may establish and defend territories soon after emerging (Shapalov and Taft 1954). Fry and juvenile steelhead that are unsuccessful in establishing a territory may suffer density-dependent mortality or be displaced downstream where they may suffer higher rates of mortality from predation, entrainment, or elevated water temperatures (Dambacher 1991, Peven et al. 1994, Reedy 1995). Keeley (2001) found that increased competition between juvenile steelhead, caused by higher fish densities or lower food densities, caused increased mortality, lower or more variable growth rates, and emigration of smaller fish. Downstream dispersal due to density dependence or high flows in rearing habitat does not necessarily result in increased mortality where there is suitable habitat downstream (Kahler et al. (2001). Downstream dispersal to larger stream reaches for further rearing prior to smolting appears common in many systems (Bjornn 1978, Loch et al. 1985, Leider et al. 1986, Dambacher 1991).

Summer habitat can generally be assumed to be more limiting for age 1+ and 2+ juvenile steelhead than for age 0+ in many streams. Older age classes of juvenile steelhead (ages 1+ and 2+) prefer deeper water in the summer than fry, and show a stronger preference for pool habitats, especially deep pools near the thalweg with ample cover, as well as higher-velocity rapid and cascade habitats (Bisson et al. 1982, 1988; Dambacher 1991). Dambacher (1991) observed that most 1+ steelhead in the Steamboat Creek watershed of the North Umpqua River, Oregon were concentrated in mainstem reaches with relatively deep riffles and large substrates. Age 1+ fish typically feed in pools, especially scour and plunge pools (Fontaine 1988, Bisson et al. 1988). Age 1+ steelhead appear to avoid secondary channel and dammed pools, glides, and low-gradient riffles with mean depths less than 7.8 in (20 cm) (Fontaine 1988, Bisson et al. 1988, Dambacher 1991). Beecher et al. (1993) reported that juvenile steelhead > 3 in (75 mm) in length avoided areas with depths of less than 6 in (15 cm). Reedy (1995) indicates that age 1+ steelhead especially prefer high-velocity pool heads, where food resources are abundant, and pool tails, which provide optimal feeding conditions in summer due to lower energy expenditure requirements than the more turbulent pool heads. Fast, deep water, in addition to optimizing feeding versus energy expenditure, provides greater protection from avian and terrestrial predators (Everest and Chapman 1972).

5.4.3 Winter rearing habitat

For juvenile steelhead to survive the winter, they must avoid predation and high flows. The higher-gradient reaches typically used for spawning by steelhead (generally > 3%) are often confined and characterized by coarse substrate that is immobile at all but the highest flows. Juvenile steelhead often use the interstitial spaces between cobbles and boulders as cover from high water velocity, and presumably, to avoid predation (Bjornn 1971, Hartman 1965, Bustard and Narver 1975, Swales et al. 1986, Everest et al. 1986, Grunbaum 1996). Access deep into the streambed may be required to avoid turbulent conditions near the surface or even beneath the first layer of the subsurface (Stillwater Sciences, unpubl. data). Age 0+ steelhead can use shallower habitats and can find interstitial cover in gravel-size substrates, while age 1+ or 2+ steelhead, because of their larger size, need coarser cobble/boulder substrate for cover (Bustard and Narver 1975; Bisson et al. 1982, 1988; Fontaine 1988; Dambacher 1991). Bustard and Narver (1975) reported that 1+ steelhead prefer water deeper than 17.5 in (45 cm) in winter, while age 0+ steelhead often occupy water less than 5.8 in (15 cm) deep and are rarely found at depths over about 23.4 in (60 cm). In winter, age 1+ steelhead typically stay within the area of streambed that remains inundated at summer low flows, while age 0+ fish frequently overwinter beyond the summer low flow perimeter along the stream margins (Everest et al. 1986). Consequently, winter rearing habitat for age 1+ and 2+ juvenile steelhead is assumed to be more limiting than for age 0+ juveniles.

5.5 Conceptual Model of Historical Habitat Conditions

Steelhead likely migrated the farthest upstream of all anadromous salmonid species in the Central Valley. Their superior jumping ability and migration during high flows probably enabled them to navigate past obstacles that may have impeded winter-run and spring-run Chinook salmon, which also ascended the high-elevation reaches of Sacramento River tributaries (Yoshiyama et al. 1996). Greater access to upstream reaches probably enabled steelhead to spawn and rear where there was less competition from spring- and winter-run Chinook salmon than in downstream reaches.

Because spawning gravels in higher-gradient reaches are often more patchily distributed than in lower-gradient reaches, steelhead likely spawned in small riffles located between steep reaches of channel and in pockets of gravel located behind boulders and LWD. Late snowmelt and volcanic springs supplied cold water to these upstream reaches throughout the summer, thus providing suitable rearing conditions throughout the summer months. Once fry emerged from the gravels, they probably migrated to nearby gravel riffles to establish and defend territories, which caused some fry to move farther downstream once rearing habitat was saturated upstream. As they dispersed downstream, steelhead fry may have encountered greater predation pressure and competition for rearing habitat from larger spring-run Chinook salmon juveniles that had established territories after emerging months earlier, and from 1+ and 2+ juvenile steelhead that had established territories in previous years. Thus, the availability of summer rearing habitat likely exerted a control on historical steelhead population abundance by limiting fry production.

Winter rearing habitat may have limited steelhead populations as well, because juveniles probably competed for limited velocity refugia during high flow events in the winter and spring. In the steep channels of higher elevation streams, juvenile steelhead find velocity cover in eddy zones associated with LWD or in the interstices between coarse sediment particles to avoid downstream displacement. Because smaller juvenile steelhead can presumably find cover in a wider range of particle sizes than larger 1+ and 2+ juveniles, and can use much shallower habitats than larger juveniles, winter habitat may be more likely to be limiting for these older age classes.

Hydrologic and geologic variability in the tributaries of the Sacramento River likely contributed to the life history flexibility displayed by steelhead. As described above, high flow events may cause age 0+ and 1+ juveniles to move downstream to lower reaches, while other juveniles took advantage of available velocity refugia to spend an additional year rearing in their natal stream before emigrating as 2+ (or occasionally 3+) juveniles.

We assume that rearing habitat for age 1+ and 2+ steelhead is likely limiting populations of steelhead in the Sacramento River system. In contrast to juvenile Chinook, which are frequently found in schools, juvenile steelhead are strongly territorial (Everest and Chapman 1972, Hillman et al. 1987). Several studies support the hypothesis that density dependence acts on the parr-to-smolt life stage rather than the egg-to-fry life stage, and that it is rearing habitat capacity that limits population size of steelhead (Bjornn 1978, Cramer et al. 1985, Ward and Slaney 1993, Cramer 2001). This is evidenced by studies showing the number of age 0+ juvenile steelhead to vary substantially over the years, while the yearly abundance of age 1+ or 2+ juveniles remains relatively stable (Bjornn 1978, Everest et al. 1987, Ward and Slaney 1993, Reeves et al. 1997). Both hatchery as well as field studies have shown that smolt-to-adult survival increases with smolt size (Shapovalov and Taft 1954, Ward et al. 1989), which emphasizes the importance of providing habitat for older age classes of juvenile steelhead.

5.6 Effects of Anthropogenic Changes on *O. mykiss* Habitat

Native Americans harvested Chinook salmon and steelhead as a food staple, and tribes located in the upper Sacramento River were particularly dependent on anadromous salmonid runs to provide sufficient food resources. However, the larger scale anthropogenic changes that have occurred in the past 150 years in the Sacramento River basin produced more significant effects on anadromous fish populations. This section describes some of the more significant anthropogenic changes to the landscape that likely had negative effects on steelhead populations in the basin.

5.6.1 Gold mining

Because steelhead ascended to the upper reaches of Sacramento River tributaries, both resident and anadromous forms of the species were often located near mining camps that were established throughout the Sierra Nevada range, thus supplying the camps with a food staple. However, alteration of aquatic habitats likely had a greater effect on *O. mykiss* populations than angling. Miners often re-routed flows from natural channels and increased sediment delivery to channels, thus degrading spawning and rearing habitat. Flow diversion may have created new flow-related passage barriers to adult upstream migration, and it probably exacerbated summer rearing habitat limitations by reducing the extent of inundated habitat, and increasing competition for limited space. In addition to reducing juvenile survival, flow diversion may have simulated the effects of drought conditions and forced *O. mykiss* juveniles to become residents to survive low flow and elevated water temperature conditions.

5.6.2 Early commercial fishing

In the middle- to late-nineteenth century, several fish canneries began operating in the lower Sacramento River and Delta to harvest the abundant salmon resources of the Central Valley. These early commercial fishing operations often used barriers and gill nets that spanned the width of channels in the Delta and the Sacramento River, effectively creating a seasonal barrier that prevented the upstream migration of anadromous species (Clark 1929). The upstream migration of steelhead generally overlaps with that of fall-run Chinook salmon, so steelhead were likely effected by the fishing operations, and they may have been a targeted species. The barriers likely caused only a partial blockage of upstream migration, because the fish racks and nets often had holes and seams that allowed individuals to pass (Hallock et al. 1961); nevertheless, early commercial fishing likely reduced steelhead escapements in the Central Valley. As Chinook salmon populations began to plummet in the late nineteenth century, the California Fish Commission began implementing angling restrictions (e.g., seasonal closures, gear restrictions) that likely reduced harvest mortality.

5.6.3 ACID Dam

The construction of the ACID Dam (RM 298.4) near Redding in 1916 likely caused delays in the upstream migration of adult steelhead, which may have caused the peak of spawning activity to shift. ACID Dam was operated seasonally, typically between April and October, so the flashboards were often in place during the beginning and the peak of steelhead upstream migration in late September. Though the delays may not have caused direct mortality, they may have contributed indirectly to adult mortality by exposing spawners to increased angling pressure as they congregated downstream of the dam. Following construction of the dam, observers noted lower escapements of Chinook salmon in the upper reaches of the Sacramento River and its tributaries (Yoshiyama et al. 1996), so the dam may have also reduced the steelhead population. The ACID Dam diversion was unscreened for many years, so juvenile steelhead migrating downstream may have been entrained in the ACID canal; however, juvenile steelhead likely suffered less entrainment mortality than juvenile winter-run Chinook salmon, because they emigrate as larger juveniles with better ability to avoid entrainment.

5.6.4 Keswick and Shasta dams

The construction of large water supply dams in the Central Valley has probably had the greatest impact on *O. mykiss* populations because it eliminated access to nearly 80% of historical spawning and rearing habitat (Lindley et al. 2006). The construction of Shasta and Keswick dams eliminated access to many of the upstream tributaries (e.g., McCloud River, Pit River, little Sacramento River) that provided the cold water temperatures required for year-round rearing by steelhead. Dam construction also landlocked potentially anadromous *O. mykiss* populations in the upper watershed, forcing them to adopt a resident life history strategy (McEwan 2001).

Though steelhead generally prefer to spawn in the higher-gradient, more confined channels associated with smaller tributaries, they will spawn in mainstem river channels; however, it is unlikely that steelhead used to spawn in the reach of the mainstem Sacramento River below Keswick Dam where they currently spawn because summer water temperatures in this reach were likely too high to support oversummering by juveniles.

Shasta Dam altered the water temperature regime of the Sacramento River, which made it possible for steelhead to spawn and rear below Keswick Dam, but it is unclear how the substitution of spawning and rearing habitats in the mainstem channel for those in the upstream tributaries affected steelhead populations. Section 4.2 described how Shasta Dam may have initially increased the amount of spawning habitat available for winter-run Chinook salmon by reducing temperatures, until bed coarsening eventually reduced spawning gravel suitability. Steelhead may have experienced similar initial increases in spawning habitat downstream of Shasta Dam, which may have compensated for some of the upstream habitat lost when the dam was completed, but this increase in spawning habitat may not have resulted in increased escapements. Steelhead employ a different life history strategy than winter-run Chinook salmon, with juveniles typically rearing in fresh water for two years before emigrating to the ocean. As juvenile steelhead establish and defend territories, rearing habitat becomes saturated, which displaces other juveniles downstream where the risk of mortality from predation, entrainment, and elevated water temperatures increases. As a result, rearing habitat, rather than spawning habitat, is more likely to be a limiting factor for steelhead, and Shasta Dam eliminated access to more summer and winter rearing habitat than was created downstream of the dam by changes in temperature regimes. Habitat modeling conducted by Lindley et al. (2006) reinforce the idea that more rearing habitat was lost than gained by indicating that Shasta Dam eliminated access to a substantial amount of rearing habitat in the McCloud, Pit, and Little Sacramento River drainages. The dam likely reduced winter rearing habitat as well by eliminating access to the cobble-bedded reaches of the upstream tributaries that provided more velocity refugia for larger juveniles during high flow events.

5.6.5 Hatchery production

Hatchery production of steelhead is very large compared to natural production, based on the Chipps Island trawl data (Good et al. 2005). The bulk of hatchery releases in the Central Valley occur in the Sacramento River basin. An analysis of steelhead captures from trawl data by Nobriga and Cadrett (2001, as cited in Good et al. 2005) indicated that hatchery steelhead comprised 63–77% of the steelhead catch. Steelhead stocks at the Mokelumne River Hatchery and Nimbus Hatchery on the American River are not part of the California Central Valley steelhead DPS due to the source of broodstock used and genetic similarities to Eel River stocks (Good et al. 2005). There are many uncertainties regarding how the hatchery programs affect the Central Valley steelhead DPS (NMFS 2006)

5.7 Management Implications, Key Hypotheses, and Uncertainties

We hypothesize that rearing habitat is the primary limiting factor for steelhead populations in the mainstem Sacramento River, especially for older age classes of juveniles (age 1+ and 2+).

Because steelhead fry require low-velocity shallow habitat upon emergence (Hartman 1965, Everest et al. 1986, Fontaine 1988), and because an average female might lay 5,500 eggs, the number of age 0+ steelhead that a reach of stream can support is small relative to the number of eggs that may be deposited, even under conditions of low escapement or high egg mortality. However, it is generally accepted that it is rearing habitat for the older age classes (age 1+ and 2+) that is usually limiting steelhead populations, as they have narrower habitat requirements. Although habitat for age 2+ juvenile steelhead is likely suitable for age 1+ juveniles, the reverse may not be true, as the older and larger juveniles may require deeper habitats in the summer and larger interstitial habitats for overwinter survival. Therefore, we hypothesize that it is the amount of summer and winter rearing habitat available to age 2+ juveniles that most likely ultimately limits the production of adult steelhead.

Trap-and-haul operations have often been criticized as an artificial and flawed method for managing fishery stocks, but it is an improvement over hatchery supplementation practices because it allows fish to spawn and rear under natural conditions and presumably reduces domestication selection.

5.7.1 Provide rearing habitat with cobble-boulder structures

As discussed above, we hypothesize that rearing habitat to support age 2+ (and possibly older) juvenile steelhead is likely the limiting factor for populations of steelhead in the Sacramento River. To expand summer and winter rearing habitat for these older juvenile steelhead, we recommend placing cobble-boulder structures in the upper Sacramento River at locations between Keswick Dam (RM 302) and Clear Creek (RM 290). The cobble-boulder structures can be placed near highways (e.g., bridge piers) and water supply structures (e.g., diversion points) in the upper Sacramento River to provide the added benefit of protecting infrastructure from channel incision and erosion. However, it is important to place cobble-boulder structures in locations where the river is not expected to meander (i.e., where the channel is confined) in order to prevent future conflicts with restoring ecological processes.

Chapter 4 described how the channel bed of the upper Sacramento River has been coarsening as a function of continued high flow releases from Shasta Dam combined with reduced sediment supply from the upper watershed. However, a channel bed that is coarsening does not necessarily mean that rearing habitat for age 2+ steelhead has been increasing as the percentage of cobble covering the bed surface has increased. The coarsened bed of the upper Sacramento River is also embedded, with gravels filling the interstitial spaces between cobbles that are used as cover by juvenile steelhead. This filling of interstitial spaces can affect habitat for all age classes of juvenile steelhead, but because of the larger size of age 1+ and older juveniles, their habitat will be reduced at lower levels of embeddedness than for age 0+ steelhead that can make use of smaller crevices. The extent to which steelhead may use riprap as cover in the Sacramento River is unknown, but its use in the Sacramento River has been documented (Schaffter et al. 1983). Lister et al. (1995) found steelhead to prefer banks protected by large, coarse riprap to those stabilized with smaller materials.

Other potential restoration measures include more significant gravel augmentation in the upper Sacramento River, coupled with flow releases that mobilize the bed periodically, which may help to create the interstices between individual sediment grains that juvenile steelhead require for cover. Another potential measure includes ripping the coarse surface layer, coupled with high flow releases, to expose coarse sediment stored in the channel subsurface to transport, which may help establish larger areas of clean gravel and gravel-cobble in downstream reaches.

5.7.2 Water temperature compliance point

As discussed in Section 5.4, moving the water temperature compliance point designed to protect winter-run Chinook salmon redds upstream from Bend Bridge (RM 258) to Balls Ferry (RM 276) could reduce the amount of spawning and rearing habitat available for late-fall-run Chinook salmon in the mainstem Sacramento River. Steelhead would likely experience loss of habitat because juveniles also rear in the mainstem river throughout the summer. However, as discussed above, we hypothesize that summer and rearing habitat to support age 2+ juveniles is likely the primary limiting factor for steelhead in the Sacramento River, but we do not know how they respond during the summer when cold water temperatures are retreating upstream. Nevertheless, the improved swimming ability and thermal tolerance of age 2+ steelhead, relative to 0+ late-fall-run Chinook oversummering in the river, suggests that the steelhead population would likely be less affected than the late-fall-run population. Consequently, future analyses of moving the water temperature compliance point to protect winter-run Chinook salmon redds should focus on the effects on the late-fall-run Chinook salmon population.

5.8 References

Bailey, E. D. 1954. Time pattern of 1953-54 migration of salmon and steelhead into the upper Sacramento River. Unpublished report. California Department of Fish and Game.

Beecher, H. A., T. H. Johnson, and J. P. Carleton. 1993. Predicting microdistributions of steelhead (*Oncorhynchus mykiss*) parr from depth and velocity preference criteria: test of an assumption of the Instream Flow Incremental Methodology. Canadian Journal of Fisheries and Aquatic Sciences 50: 2380-2387.

Bell, M. C., editor. 1986. Fisheries handbook of engineering requirements and biological criteria. Report No. NTIS AD/A167-877. Fish Passage Development and Evaluation Program, U.S. Army Corps of Engineers, North Pacific Division, Portland, Oregon.

Bisson, P. A., K. Sullivan, and J. L. Nielsen. 1988. Channel hydraulics, habitat use, and body form of juvenile coho salmon, steelhead trout, and cutthroat trout in streams. Transactions of the American Fisheries Society 117: 262-273.

Bisson, P., J. L. Nielsen, R. A. Palmason, and L. E. Grove. 1982. A system of naming habitat types in small streams, with examples of habitat utilization by salmonids during low streamflows. Pages 62-73 in N. B. Armantrout, editor. Proceedings of the symposium on acquisition and utilization of aquatic habitat inventory information. American Fisheries Society, Western Division, Bethesda, Maryland.

Bjornn, T. C. 1971. Trout and salmon movements in two Idaho streams as related to temperature, food, stream flow, cover, and population density. Transactions of the American Fisheries Society 100: 423-438.

Bjornn, T. C. 1978. Survival, production, and yield of trout and chinook salmon in the Lemhi River, Idaho. Bulletin No. 27. Prepared by Idaho Cooperative Fishery Research Unit, College of Forestry, Wildlife and Range Sciences, University of Idaho, Moscow for Idaho Department of Fish and Game.

Bjornn, T. C. and D. W. Reiser. 1991. Habitat requirements of salmonids in streams. American Fisheries Society Special Publication 19:83-138.

Bovee, K.D. 1978. Probability-of-use criteria for the family Salmonidae. Instream Flow Information Paper 4, U.S. Fish and Wildlife Service, FWS/OBS-78/07. 79pp.

Burnett, K. M. 2001. Relationships among juvenile anadromous salmonids, their freshwater habitat, and landscape characteristics over multiple years and spatial scales in the Elk River, Oregon. Doctoral dissertation. Oregon State University, Corvallis, Oregon.

Bustard, D. R., and D. W. Narver. 1975. Aspects of the winter ecology of juvenile coho salmon (*Oncorhynchus kisutch*) and steelhead trout (*Salmo gairdneri*). Journal of the Fisheries Research Board of Canada 32: 667-680.

CH2M Hill. 1985. Klamath River basin fisheries resource plan. Prepared for U.S. Department of the Interior.

Chambers, J. S., R. T. Pressey, J. R. Donaldson, and W. R. McKinley. 1954. Research relating to study of spawning grounds in natural areas. Annual Report, Contract No. DA 35026-Eng-20572. Prepared by Washington State Department of Fisheries, Olympia for U.S. Army Corps of Engineers, Fisheries-Engineering Research Program, North Pacific Division, Portland, Oregon.

Chambers, J. S., G. H. Allen, and R. T. Pressey. 1955. Research relating to study of spawning grounds in natural areas. Annual Report, Contract No. DA 35026-Eng-20572. Prepared by Washington State Department of Fisheries, Olympia for U.S. Army Corps of Engineers, Fisheries-Engineering Research Program, North Pacific Division, Portland, Oregon.

Clark, G. H. 1929. Sacramento-San Joaquin salmon (*Oncorhynchus tshawytscha*) fishery of California. California Department of Fish and Game Fish Bulletin 17: 1-73

Cramer, S. P. 2001. The relationship of stream habitat features to potential for production of four salmonid species. Prepared by S. P. Cramer and Associates, Gresham, Oregon for Oregon Building Industry Association.

Cramer, S. P., D. W. Alley, J. E. Baldrige, K. Barnard, D. B. Demko, D. H. Dettman, B. Farrell, J. Hagar, T. P. Keegan, A. Laird, W. T. Mitchell, R. C. Nuzum, R. Orton, J. J. Smith, T. L. Taylor, P. A. Unger, and E. S. Van Dyke. 1995. The status of steelhead populations in California in regards to the Endangered Species Act. Special report submitted to National Marine Fisheries Service on behalf of Association of California Water Agencies. S. P. Cramer & Associates, Gresham, Oregon.

Cramer, S. P., T. D. Satterthwaite, R. B. Boyce, and B. P. McPherson. 1985. Impacts of the Lost Creek Dam on the biology of anadromous salmon in the Rogue River. Report No. DACW57-77-C-0027. Prepared by Oregon Department of Fish and Game, Research and Development Section for U.S. Army Corps of Engineers.

Dambacher, J. M. 1991. Distribution, abundance, and emigration of juvenile steelhead (*Oncorhynchus mykiss*), and analysis of stream habitat in the Steamboat Creek basin, Oregon. Master's thesis. Oregon State University, Corvallis.

Engle, R. O. 2002. Distribution and summer survival of juvenile steelhead trout (*Oncorhynchus mykiss*) in two streams within the King Range National Conservation Area, California. Master's thesis. Humboldt State University, Arcata, California.

Everest, F. H., and D. W. Chapman. 1972. Habitat selection and spatial interaction by juvenile chinook salmon and steelhead trout in two Idaho streams. Journal of the Fisheries Research Board of Canada 29: 91-100.

Everest, F. H., G. H. Reeves, J. R. Sedell, J. Wolfe, D. Hohler, and D. A. Heller. 1986. Abundance, behavior, and habitat utilization by coho salmon and steelhead trout in Fish Creek, Oregon, as influenced by habitat enhancement. Annual Report 1985 Project No. 84-11. Prepared by U.S. Forest Service for Bonneville Power Administration, Portland, Oregon.

Everest, F. H., G. H. Reeves, J. R. Sedell, and D. B. Hohler. 1987. The effects of habitat enhancement on steelhead trout and coho salmon smolt production, habitat utilization, and habitat availability in Fish Creek, Oregon, 1983-86. Annual Report 1986, Contract DE-A179-BP16726, Project 84-11. Prepared for Bonneville Power Administration, Portland, Oregon.

Fontaine, B. L. 1988. An evaluation of the effectiveness of instream structures for steelhead trout rearing habitat in the Steamboat Creek basin. Master's thesis. Oregon State University, Corvallis.

Good, T. P., R. S. Waples, and P. Adams. 2005. Updated status of federally listed ESUs of west coast salmon and steelhead. NOAA Technical Memorandum NMFS-NWFSC-66. National Marine Fisheries Service, Northwest Fisheries Science Center, Seattle, Washington and NMFS, Southwest Fisheries Science Center, Santa Cruz, California.

Grunbaum, J. B. 1996. Geographical and seasonal variation in diel habitat use by juvenile (age 1+) steelhead trout (*Oncorhynchus mykiss*) in Oregon coastal and inland streams. Master's thesis. Oregon State University, Corvallis.

Hallock, R. J., W. F. Van Woert, and L. Shapovalov. 1961. An evaluation of stocking hatchery-reared steelhead rainbow trout (*Salmo gairdnerii gairdnerii*) in the Sacramento River system. Fish Bulletin 114. California Department of Fish and Game.

Hartman, G. F. 1965. The role of behavior in the ecology and interaction of underyearling coho salmon (*Oncorhynchus kisutch*) and steelhead trout (*Salmo gairdneri*). Journal of the Fisheries Research Board of Canada 22: 1035-1081.

Harvey, B. C., J. L. White, and R. J. Nakamoto. 2002. Habitat relationships and larval drift of native and nonindigenous fishes in neighboring tributaries of a coastal California river. Transactions of the American Fisheries Society 131: 159-170.

Hicks, B. J., and J. D. Hall. 2003. Rock type and channel gradient structure salmonid populations in the Oregon Coast Range. Transactions of the American Fisheries Society 132: 468-482.

Hillman, T. W., J. S. Griffith, and W. S. Platts. 1987. Summer and winter habitat selection by juvenile chinook salmon in a highly sedimented Idaho stream. Transactions of the American Fisheries Society 116: 185-195.

Hooper, D. R. 1973. Evaluation of the effects of flows on trout stream ecology. Pacific Gas and Electric Company, Department of Engineering Research, Emeryville, California.

Kahler, T. H., P. Roni, and T. P. Quinn. 2001. Summer movement and growth of juvenile anadromous salmonids in small western Washington streams. Canadian Journal of Fisheries and Aquatic Sciences 58: 1947-1956.

Keeley, E. R. 2001. Demographic responses to food and space competition by juvenile steelhead trout. Ecology 82: 1247-1259.

Kondolf, G. M., and M. G. Wolman. 1993. The sizes of salmonid spawning gravels. Water Resources Research 29: 2275-2285.

Leider, S. A., M. W. Chilcote, and J. J. Loch. 1986. Movement and survival of presmolt steelhead in a tributary and the main stem of a Washington river. North American Journal of Fisheries Management 6: 526-531.

Lindley, S.T., R.S. Schick, A. Agrawal, M. Goslin, T.E. Pearson, E. More, J.J. Anderson, B. May, S. Greene, C. Hanson, A. Low, D. McEwan, R.B. MacFarlane, C. Swanson, and J.G. Williams. 2006. Historical Population Structure of Central Valley Steelhead and its Alteration by Dams, San Francisco Estuary and Watershed Science, Vol. 4, Issue 1: 1-19.

Lister, D. B., R. J. Beniston, R. Kellerhals, and M. Miles. 1995. Rock size affects juvenile salmonid use of stream bank riprap. Pages 621-632 in C. R. Thorne, S. R. Abt, F. J. B. Barends, S. T. Maynord and K. W. Pilarczyk, editors. River, coastal and shoreline protection: erosion control using riprap and armourstone. John Wiley and Sons, New York.

Loch, J. J., M. W. Chilcote, and S. A. Leider. 1985. Kalama River studies final report. Part II: Juvenile downstream migrant studies. Fish Management Division Report No. 85-12. Washington Department of Fish and Game, Olympia.

McEwan, D. and T.A. Jackson. 1996. Steelhead Restoration and Management Plan for California. California Department of Fish and Game.

McEwan, D. 2001. Central Valley Steelhead. In Contributions to the Biology of Central Valley Salmonids, Fish Bulletin 179. California Dept. of Fish and Game.

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

Myrick, C. A., and Jr. J. J. Cech. 2001. Temperature effects on chinook salmon and steelhead: a review focusing on California's Central Valley populations. Prepared by Department of Fishery and Wildlife Biology, Colorado State University, Fort Collins and Department of Wildlife, Fish, and Conservation Biology, University of California, Davis for the Bay-Delta Modeling Forum. http://www.sfei.org/modelingforum/.

Needham, P. R., O. R. Smith, and H. A. Hanson. 1941. Salmon salvage problems in relation to Shasta Dam, California, and notes on the biology of the Sacramento River salmon. Transactions of the American Fisheries Society 70: 55-69.

NMFS (National Marine Fisheries Service). 1998. Endangered and threatened species; threatened status for two ESUs of steelhead in Washington, Oregon, and California. Federal Register 63: 13347-13371.

NMFS. 2006. Endangered and threatened species: final listing determinations for 10 Distinct Population Segments of west coast steelhead. Federal Register 71: 834-862.

Nobriga, M., and P. Cadrett. 2001. Differences among hatchery and wild steelhead: evidence from delta fish monitoring programs. Interagency Ecological Program for the Sacramento-San Joaquin Estuary Newsletter 14: 30-38.

Peven, C. M., R. R. Whitney, and K. R. Williams. 1994. Age and length of steelhead smolts from the mid-Columbia River basin, Washington. North American Journal of Fisheries Management 14: 77-86.

Platts, W. S., M. A. Shirazi, and D. H. Lewis. 1979. Sediment particle sizes used by salmon for spawning with methods for evaluation. Ecological Research Series EPA-600/3-79-043. U.S. Environmental Protection Agency, Corvallis Environmental Research Laboratory, Corvallis, Oregon.

Reedy, G. D. 1995. Summer abundance and distribution of juvenile chinook salmon (*Oncorhynchus tshawytscha*) and steelhead trout (*Oncorhynchus mykiss*) in the Middle Fork Smith River, California. Master's thesis. Humboldt State University, Arcata, California.

Reiser, D. W., and T. C. Bjornn. 1979. Habitat requirements of anadromous salmonids. Pages 1-54 *in* W. R. Meehan, editor. Influence of forest and rangeland management on anadromous fish habitat in western North America. General Technical Report PNW-96. USDA Forest Service, Pacific Northwest Forest and Range Experiment Station, Portland, Oregon.

Schaffter, R. G., P. A. Jones, and J. G. Karlton. 1983. Sacramento River and tributaries bank protection and erosion control investigation, evaluation of impacts on fisheries. Final Report. California Department of Fish and Game.

Shapovalov, L., and A. C. Taft. 1954. The life histories of the steelhead Rainbow trout (*Salmo gairdneri gairdneri*) and silver salmon (*Oncorhynchus kisutch*) with special reference to Waddell Creek, California, and recommendations regarding their management. DFG Bulletin No. 98.

Swales, S., R. B. Lauzier, and C. D. Levings. 1986. Winter habitat preferences of juvenile salmonids in two interior rivers in British Columbia. Canadian Journal of Zoology 64: 1506-1514.

USBR (U.S. Bureau of Reclamation). 2004. Long-term Central Valley Project and State Water Project Operations Criteria and Plan Biological Assessment. USDI Bureau of Reclamation, Mid-Pacific Region, Sacramento, California.

USFWS and CDFG (U.S. Fish and Wildlife Service and California Department of Fish and Game). 1953. A plan for the protection and maintenance of salmon and steelhead in the American River, California, together with recommendations for action. Unpublished report.

Voight, H. N., and D. B. Gale. 1998. Distribution of fish species in tributaries of the lower Klamath River: an interim report, FY 1996. Technical Report No. 3. Yurok Tribal Fisheries Program, Habitat Assessment and Biological Monitoring Division.

Ward, B. R., and P. A. Slaney. 1993. Egg-to-smolt survival and fry-to-smolt density dependence of Keogh River steelhead. Pages 209-217 *in* R. J. Gibson and R. E. Cutting, editors. Production of juvenile Atlantic salmon, *Salmo salar*, in natural waters. Canadian Special Publication of Fisheries and Aquatic Sciences 118.

Ward, B. R., P. A. Slaney, A. R. Facchin, and R. W. Land. 1989. Size-biased survival in steelhead trout (*Oncorhynchus mykiss*): back-calculated lengths from adults' scales compared to migrating smolts at the Keogh River, British Columbia. Canadian Journal of Fisheries and Aquatic Sciences 46: 1853-1858.

Yoshiyama RM, Gerstung ER, Fisher FW, Moyle PB. 1996. Historical and present distribution of Chinook salmon in the Central Valley drainage of California. In: Sierra Nevada ecosystem project: final report to Congress. Volume III: assessments, commissioned reports, and background information. Davis (CA): University of California, Centers for Water and Wildlife Resources. p 309–61.

Yoshiyama, R. M., E. R. Gerstung, F. W. Fisher, and P. B. Moyle. 2001. Historical and present distribution of chinook salmon in the Central Valley drainage of California. Pages 71-176 in R. L. Brown, editor. Contributions to the biology of Central Valley salmonids. Fish Bulletin 179: Volume 1. California Department of Fish and Game, Sacramento.

6 NORTH AMERICAN GREEN STURGEON

The North American green sturgeon (*Acipenser medirostros*) is a large, long-lived anadromous species that lives most of its life in marine environments, often migrating thousands of miles along the western coast of the United States and Canada. Adults migrate periodically into freshwater streams to spawn, and the Sacramento River supports one of only three known spawning populations, in addition to the Rogue and Klamath rivers in Oregon. The NMFS has divided North American green sturgeon into two Distinct Population Segments (DPS) using the Eel River in California as the line of demarcation (Adams et al. 2002). The population that spawns in the Sacramento River constitutes the only known population in the Southern DPS, which NMFS listed as a threatened under the federal ESA in April, 2006 (NFMS 2006).

Green sturgeon were selected as a focal species, despite the paucity of specific habitat information in the Sacramento River, because the recent listing of the southern DPS highlights potential conflicts with existing resource management practices (e.g., flow releases, water temperature compliance points, diversion dam operations). As part of the consideration to list green sturgeon, NMFS and others have compiled excellent reviews of existing information about the species, including general habitat preferences and life history timing that can be applied to the southern DPS to compensate for the lack of site-specific data (EPIC et al. 2001; Adams et al. 2002; NMFS 2005; Beamesderfer et al. 2004). This chapter attempts to build on these excellent reviews by exploring potential protection and restoration measures for green sturgeon in the Sacramento River.

6.1 Geographic Distribution and Population Trends

6.1.1 Geographic distribution of North American green sturgeon

North American green sturgeon are the most wide-ranging of the sturgeon species, with ocean migrations that range between northern Mexico and southern Alaska (Adams et al. 2002). Ocean abundance and densities of green sturgeon increase north of the Golden Gate, because both the southern DPS and northern DPS generally migrate northward along the coast (NMFS 2005) as confirmed by radio telemetry studies conducted on Sacramento River green sturgeon (CDFG 2002, as cited in Adams et al. 2002). Sub-adult and adult green sturgeon migrate thousands of miles along the western coast of the United States, often venturing into coastal estuaries like Willapa Bay and Greys Harbor in Washington, where they concentrate during the summer (Adams et al. 2002). Two adults tagged in Willapa Bay have been detected by radio telemetry stations in the Sacramento River (J. Heublein, personal communication, November 14, 2005), indicating that green sturgeon from the Sacramento River migrate as far north as Washington before returning to the Sacramento to spawn. Concentrations of green sturgeon have also been detected near Vancouver Island in Canada (NMFS 2005). The long-distance ocean migrations and the residence time of sub-adults and adults in estuaries along the coast, where they are subject to both recreational and commercial fishing pressure, indicate that factors outside of the Sacramento River affect the spawning population of the southern DPS.

Though green sturgeon migrate thousands of miles through rivers, estuaries, and ocean during their long lives, they do not appear to readily establish new sub-populations, as evidenced by the documentation of only three known spawning populations in the Sacramento, Rogue, and Klamath river systems. However, recent data suggests the possibility of spawning populations in both the Eel River, California, and the Umpqua River, Oregon (NMFS 2005), which could indicate the presence of a previously undetected remnant population or the seeds of a new sub-population.

6.1.2 Distribution within the Sacramento River

Though green sturgeon spawning has been documented in the Sacramento River, available data only supports a rough approximation of spawning locations. Larval green sturgeon have been captured routinely, but in very small numbers in the RBDD rotary screw trap (RST) (RM 243.5) and the GCID fish facility (RM 206), suggesting that spawning generally occurs upstream of Hamilton City (RM 199), though some researchers believe that spawning may occur as far downstream as Chico Landing (RM 194) (J. Heublein, personal communication, November 14, 2005). Agency personnel have observed adult green sturgeon congregating below RBDD during the late spring and early summer when the gates are down (Beamesderfer et al. 2004), suggesting that they may be ripe adults trying to migrate upstream to spawn. Spawning may occur in reaches upstream of the dam (CDFG 2002, as cited in Adams et al. 2002), but the upstream extent of spawning is unknown. In 1999, the United States Fish and Wildlife Service (USFWS) placed egg mats in the Sacramento River from ACID Dam (RM 298.4) to a reach ten miles downstream of RBDD to identify green sturgeon spawning sites. However, only 2 eggs were captured, both at mats located downstream of RBDD, so the study did not clarify the location of specific spawning sites or the upstream extent of spawning (Beamesderfer et al. 2004). The UCD radio telemetry study recently detected two adult green sturgeon migrating past a remote monitoring station located above RBDD (RM 243.5) suggesting a possible spawning migration to an area upstream (J. Heublein, personal communication, November 14, 2005).

6.1.3 Population trend of the southern DPS

Population estimates of green sturgeon in the Sacramento River are derived from data collected by monitoring programs that are generally focused on other species, because there is no monitoring program targeted specifically for green sturgeon in the Sacramento River. The annual Green sturgeon larvae are captured annually in the RBDD rotary screw trap (RST), the GCID fish screen, and the CVP/SWP fish salvage facilities located downstream in the south Delta. CDFG also conducts annual trammel net surveys in San Pablo bay to track the white sturgeon population, and green sturgeon are often part of the incidental catch.

Though the annual catch of green sturgeon larvae in the RBDD RST and the GCID fish screen suggests that spawning occurs annually, the data does not permit the identification of a clear trend in abundance (Adams et al. 2002).

6.2 Life History and Habitat Requirements

6.2.1 Life history

Though green sturgeon spend the majority of their life in marine and estuarine environments, they periodically migrate into freshwater streams to spawn, spending up to six months in freshwater during their spawning migration. Upstream migration generally begins in February and may last until late July (Adams et al. 2002). In the Rogue River, upstream migrants hold in deep pools with slow velocities before migrating farther upstream to spawn (Erickson et al. 2002). Spawning occurs between March and July, peaking between mid-April and mid-June (Emmett et al. 1991, as cited in Moyle 2002). In the Rogue River, adults hold in deep pools after spawning until the late fall or early winter, when they emigrate to downstream estuaries or the ocean, perhaps cued by winter freshets that cause water temperatures to drop below 50°F (10°C) (Erickson et al. 2002). It is not known if green sturgeon exhibit this pre- and post-spawning holding behavior in the Sacramento River, though anecdotal evidence provided by anglers suggests that they do.

Larval green sturgeon grow quickly, reaching 2.9 in (74 mm) by the time they become juveniles at around 45 days post-hatching (Deng 2000). Laboratory studies suggest that optimal growth rates occur at water temperatures of 59°F (15°C), with depressed growth at water temperatures below 52°F (11°C) and above 66°F (19°C) (Cech et al. 2000). Larval green sturgeon are captured at the RBDD RST (RM 243) and the GCID facility (RM 206) between May and August, with peak capture at RBDD in June and July and at the GCID fish facility in July (RM 206) (Adams et al. 2002) (Figure 6-1).

Several studies suggest that juvenile green sturgeon rear in freshwater for 1–4 years, acclimating gradually to brackish environments before migrating to the ocean (Beamesderfer and Webb 2002, Nakamoto et al. 1995). Green sturgeon larvae trapped at RBDD (RM 243.5) have an average length of 1.1 in (2.9 cm), while larvae trapped at GCID (RM 206) have an average length of 1.4 in (3.6 cm) (Adams et al. 2002), suggesting that larvae begin moving downstream soon after hatching. However, it is not clear how long larval and juvenile green sturgeon rear in the middle Sacramento River. Klamath River studies indicate that juvenile green sturgeon can grow to 12 in (30 cm) in their first year and 24 in (60 cm) within two to three years (Nakamoto et al. 1995). The salvage of juvenile green sturgeon at the CVP and SWP fish facilities indicates that they rear in the Bay-Delta estuary (Adams et al. 2002), though it is unclear for how long before migrating to the ocean.

Sub-adult and adult green sturgeon generally migrate north along the coast once they reach the ocean, concentrating in coastal estuaries like Willapa Bay, Grays Harbor, and the Columbia River estuary in Washington during summer months (Adams et al. 2002). The strategy underlying summer visits to coastal estuaries is unclear, because sampling indicates relatively empty stomachs, suggesting that they may not enter the estuaries to feed (Beamesderfer 2000, as cited in Adams et al. 2002). Females reach sexual maturity after about 17 years, and males reach maturity after about 15 years (Adams et al. 2002). Spawning was previously believed to occur every 3–5 years (Tracy 1990, as cited in Adams et al. 2002), but may occur as frequently as every 2 years (Lindley and Moser, pers. comm., 2004, as cited in NMFS 2005).

Similar to many other anadromous fish on many other river systems, green sturgeon likely evolved to the seasonal pattern of flows, or other variables affected by flows, in the Sacramento River. However, it is not clear if anthropogenically induced changes in the flow regime have contributed to the apparent decline in green sturgeon spawners. Some researchers have hypothesized that high spring flows, or the turbidity associated with them, may provide an upstream migration cue. The annual catch of larval sturgeon at the RBDD (RM 243.5) and GCID (RM 206) screens suggests that spawning occurs in the Sacramento River in most years, regardless of water year type; however, it is unclear how many adults return to spawn each year, so it is unclear if there is a relationship between flow events and the number of adult spawners in a given year. The relationship between flow and water temperature in the Sacramento River may influence the southern DPS by determining the amount of suitable rearing habitat available for larval and juvenile green sturgeon, which typically rear in the river and Bay-Delta estuary for 1–4 years before emigrating to the estuary and ocean (Adams et al. 2002).

6.2.2 Habitat requirements

Though there are general descriptions of preferred habitat conditions for green sturgeon, much of this information is derived from the Rogue and Klamath rivers, and little is known about specific spawning, rearing, or holding habitat locations and conditions in the Sacramento River. Researchers at the University of California, Davis (UCD) are conducting radio telemetry studies on green sturgeon that may yield better information about preferred habitats in the Sacramento River in the near future (P. Klimley, personal communication, March 1, 2005).

6.2.2.1 Spawning habitat

Green sturgeon seem to prefer areas of fast, deep, turbulent water in mainstem channels as spawning habitat (Moyle 2002). They spawn in a wide variety of substrates, from clean sand to bedrock, but they appear to prefer bed surfaces composed of coarse cobble (Moyle 2002). The interstices between the large particles may allow eggs to lodge in the bed surface to provide cover from predators (Moyle 2002). In the Rogue River, suspected spawning sites (inferred from the movement of tagged green sturgeon as part of a telemetry study) have beds composed of cobbles and boulders, with water depths greater than 10–15 ft (3–4.6 m), and turbulent water associated with slope breaks in the channel (D. Erickson, personal communication, July 14, 2005).

6.2.2.2 Post-spawning holding habitat

In the Rogue River, telemetry studies have shown that adult green sturgeon hold in low-velocity, deep water habitats prior to migrating upstream to spawn. The adults move around in the pools, and may stray short distances from a pool, but the scope of their movement is limited. Following spawning, adults migrate downstream to hold in the low-velocity, deep pools through the summer and fall months until emigrating downstream to the estuary or ocean. Researchers in the Rogue River have observed post-spawning adults emigrating downstream from their holding habitat during late fall and early winter freshets. They hypothesize that water temperatures, rather than changes in flow magnitude, provide the migratory cue, because they have correlated the timing of adult emigration with water temperatures falling below 50°F (10°C) (Erickson et al. 2002).

6.2.2.3 Larval rearing habitat

The rearing habitat preferences of green sturgeon larvae and juveniles in the Sacramento River is poorly understood. Recent laboratory research has identified water temperature thresholds for larval green sturgeon (Van Eenennaam et al. 2001). Water temperatures over 68°F (20°C) were found to be lethal to green sturgeon embryos (Cech et al. 2000), and temperatures above 63–64°F (17–18°C) were found to be stressful (Van Eenennaam et al. 2005). Optimal growth of larvae occurred at 59°F (15°C), with depressed growth at water temperatures below 52°F (11°C) and above 62°F (19°C) (Cech et al. 2000).

6.3 Conceptual Model of Historical Habitat Conditions

The location and character of spawning sites in the Rogue and Klamath rivers suggests that green sturgeon likely spawned in the Sacramento River above the current site of Keswick Dam (RM 302), including the Pit, McCloud, and Little Sacramento rivers (Nakamoto et al. 1995, NMFS

2005). The upstream migration period (February through July) corresponds with winter base flows, winter flow events, and spring snowmelt. Adult green sturgeon likely entered the Sacramento River during the winter, holding in pools in the middle and upper Sacramento River until high flow events triggered upstream migration, indicating that adults could navigate flow-related passage barriers to access spawning sites farther upstream.

By migrating far upstream to spawn, adults gained access to steep channels with high flow velocities and coarse bed surfaces. As a broadcast spawner that produces eggs that are relatively non-adhesive, green sturgeon relied on the coarse bed surfaces in the upper watershed to provide interstices where eggs could lodge in crevices protected from potential predators. Green sturgeon eggs have the largest mean diameter of any sturgeon species (Cech et al. 2000), but they lay fewer eggs than other sturgeon species. The larger egg diameter provides more yolk resources for embryos, which may allow them to grow larger before emerging from cover, thereby increasing their survival relative to other sturgeon species.

Following emergence in early summer, larval green sturgeon began migrating downstream with snowmelt flows between May and July, growing quickly and becoming more tolerant of increasing water temperatures and salinities.

6.4 Effects of Anthropogenic Changes on Green Sturgeon Habitat

Little is known about the historical abundance and distribution of green sturgeon in the Sacramento River basin. Nevertheless, several anthropogenic changes have likely contributed to the decline in green sturgeon. The location and characteristics of preferred green sturgeon spawning habitats in the Rogue and Klamath rivers suggest that most of the historical spawning habitat in the Sacramento River likely occurred upstream of the current Keswick Dam site (RM 302), so that dam construction in the 1940s created a permanent barrier that eliminated the majority of available spawning habitat. Upstream passage may have been impeded even sooner by the seasonal operation of the ACID Dam near Redding (RM 298.4), which began operations in 1916. The spawning population that was displaced downstream of the dams have likely experienced further reductions in access to spawning habitat by the operation of RBDD (RM 243.5), which blocks upstream access for late migrants when the gates are lowered in mid-May. Beginning in the late nineteenth century, water diversions such as the GCID diversion (RM 206) near Hamilton City may have entrained the larvae of successful upstream migrants and spawners.

6.4.1 Commercial fishing

Commercial fishermen in the late nineteenth century used barriers and gill nets that spanned the width of channels in the Delta and the Sacramento River to trap anadromous fish migrating upstream (Clark 1929). Chinook salmon were the primary target of these fishing efforts, but green sturgeon may have constituted part of the by-catch. Spring-run Chinook salmon were a prized target of both commercial and recreational harvest because they began their upstream migration in the best physical condition and often provided the best quality meat of the four runs of Chinook salmon. The upstream migration period of green sturgeon overlaps with that of spring-run salmon, so sturgeon were likely trapped by the nets and barriers in the lower reaches of the Sacramento River and the Delta. The barriers likely caused only a partial blockage of upstream migration, because the fish racks and nets often had holes and seams that allowed individuals to pass (Hatton 1940). Nevertheless, early commercial fishing likely reduced the number of adult green sturgeon that spawned in the Sacramento River by preventing or delaying upstream migration. As Chinook salmon populations began to plummet in the late nineteenth century, the California Fish Commission began implementing angling restrictions (e.g., seasonal closures, gear restrictions) that likely benefited green sturgeon as well.

6.4.2 ACID Dam

The completion of the ACID Dam (RM 298.4) near Redding in 1916 likely impeded green sturgeon access to historical spawning habitats. Because the dam was operated seasonally, typically between April and October, it functioned as a partial barrier, allowing early upstream migrants to navigate past the site in February and March but blocking later migrants. Following construction of the dam, observers noted lower escapements of Chinook salmon in the upper reaches of the Sacramento River and its tributaries (Yoshiyama et al. 1996), and the dam likely produced similar effects on green sturgeon. Because the dam diverted unscreened water into a canal (until 1927, when the diversion was screened), it likely resulted in loss of juvenile green sturgeon migrating downstream.

6.4.3 Keswick and Shasta dams

The construction of Keswick and Shasta dams in the 1940s created a permanent barrier to upstream migration, which we hypothesize eliminated access to most of the historical spawning habitat used by green sturgeon. However, Shasta Dam likely created new spawning habitat in downstream reaches by changing the water temperature regime (NMFS 2005). The release of cold water though the summer allowed green sturgeon to spawn in these downstream reaches because larvae and juveniles could now survive through the summer months during their downstream migrations. On balance, however, the dams almost certainly eliminated more habitat than they created.

6.4.4 Upstream passage problems at RBDD

The construction and operation of RBDD in 1964 likely had significant impacts on green sturgeon spawning by blocking access to upstream spawning areas. Prior to 1986, RBDD gates were closed during most of the upstream migration period for green sturgeon. Though spawning habitat may exist between RBDD and Hamilton City (RM 199), the capture of larval green sturgeon in the RBDD Rotary Screws Trap indicates that at least some spawning occurs upstream of the dam, such that the loss of this habitat upstream of RBDD compounded the loss of historical spawning habitat caused by Keswick and Shasta dams. Gate operations at RBDD were changed in 1986 to facilitate the upstream passage of winter-run Chinook salmon, but the change in gate operations also encompassed a large percentage of the upstream migration period for adult green sturgeon. However, RBDD gates are usually lowered beginning May 15th, so RBDD operations still prevent upstream migration for the tail end of the green sturgeon migration period. RBDD may also have diverted larval green sturgeon into the Tehama-Colusa Canal. Juvenile Chinook salmon migrating downstream through the open gates at RBDD have been reported to become disoriented and vulnerable to predation from the turbulence caused by water flowing over the dam infrastructure. It is unknown if larval green sturgeon experience similar disorientation or if they are vulnerable to predation as they transit through the dam's open gates, but the issue merits study.

6.5 Conceptual Model of Current Conditions

Little is known about the specific locations and habitat used by the spawning population of green sturgeon in the Sacramento River because they have been the subject of very little study. More information about habitat use is available for white sturgeon populations in the Sacramento River basin, which has been used as a proxy for green sturgeon in the Sacramento River. However, the two species have sufficiently different distributions within the basin to suggest that they have different habitat needs and preferences. The following conceptual model of how green sturgeon use the mainstem Sacramento River draws upon information about habitat preferences derived from research conducted on the spawning populations of green sturgeon in the Rogue and Klamath river systems.

Green sturgeon begin migrating upstream in February, possibly holding in pools in the vicinity of Hamilton City. Closure of the RBDD gates in mid-May prevents late migrants from accessing upstream spawning sites, forcing them to spawn downstream or to abandon spawning altogether. Green sturgeon spawn upstream of Hamilton City (RM 199), perhaps as far upstream as Keswick Dam (RM 302) (CDFG 2002, as cited in NMFS 2003) beginning in March, and peaking between mid-April and mid-June. We hypothesize that, after spawning, adults migrate downstream to hold in pools in the middle Sacramento River until late fall or early winter storms provide a cue to migrate downstream to the estuary.

Green sturgeon larvae begin to emerge and move downstream beginning in May, with peak passage occurring at RBDD and the GCID facility in June and July. Historically, migrating downstream during the snowmelt period may have helped green sturgeon juveniles emigrate quickly to reduce their exposure to predation, and the higher discharge and associated turbidity likely helped juveniles avoid potential predators, especially visual predators. Current water temperature controls designed to protect winter-run Chinook salmon help to maintain a favorable water temperature regime in the middle Sacramento River, allowing green sturgeon larvae to grow quickly so that they transform into juveniles within 45 days of hatching. As the juveniles move through the middle and lower Sacramento River, they continue their quick growth while acclimating to increasing water temperatures and, eventually, the salinities of the estuary.

6.6 Management Implications, Key Hypotheses, and Uncertainties

6.6.1 RBDD gate operations

The listing of the southern DPS of green sturgeon as a threatened species will likely pose a management conflict with RBDD gate operations. The USBR began raising RBDD gates to facilitate upstream passage of winter-run Chinook salmon in the winter of 1986 (Table 4.2-1), which likely improved upstream access for early green sturgeon migrants moving upstream in February. Another change in gate operations occurred in 1993, when the USBR began raising the dam gates between mid-September and mid-May, which likely conferred an even greater benefit to green sturgeon because it covered more of the upstream migration window (Table 4.2-1). Nevertheless, current RBDD operations close the gates in mid-May, which clips the end of the upstream migration period for green sturgeon. Agency biologists have observed adult green sturgeon congregating downstream of RBDD when the gates are closed in the late spring and early summer, suggesting that some ripe adults are trying to access upstream spawning sites. Considering the few adults that appear to return to the Sacramento River to spawn each year, if current RBDD gate operations prevent even a small number of adult green sturgeon from accessing suitable spawning sites, the effects on the population could be significant. The current radio telemetry studies may yield better information on upstream migration timing, which will facilitate an assessment of the potential conflict between current RBDD gate operations and green sturgeon habitat needs.

6.6.2 Upstream migration cue

If the Sacramento River population of green sturgeon exhibits a pre-spawning holding phase as has been observed on the Rogue River, then flow events may provide a cue to adults to resume their upstream migration to spawn. If adult green sturgeon can be induced to migrate upstream by a flow cue, then it may be possible to reduce the conflict with current RBDD gate operations by trying to lure adults upstream prior to gate closure. If the Sacramento population does not hold in the middle Sacramento River prior to spawning, then the use of a managed flow pulse as an upstream migration cue would be ineffective, because the cue would have to extend all the way to the Bay-Delta estuary where the adults are located.

6.6.3 Water temperature compliance point

As described in Chapter 4.2, the USBR currently operates the Shasta-Keswick and Trinity divisions of the CVP to maintain water temperatures at 56^{o}F (13^{o}C) down to Bend Bridge (RM 258) between April 15 and September 30, as required by a 1993 NMFS Biological Opinion (NMFS 1993). The USBR has petitioned to move the water temperature compliance point upstream to Balls Ferry (RM 276) to facilitate greater flexibility in managing the coldwater pool in Shasta Lake for winter-run salmon.

Chapter 4.5 explained that moving the water temperature compliance point upstream would likely have negative effects on the late-fall-run salmon population by reducing the amount of summer rearing habitat available to fingerlings that use the yearling life history strategy. Because green sturgeon larvae and juveniles rear in freshwater for 1–3 years before emigrating to the estuary, moving the temperature compliance point upstream could have similar deleterious effects on green sturgeon juveniles. Chapter 4.5 also argues that summer water temperatures may dictate the location of late-fall-run salmon spawning by controlling where late-fall-run juveniles are able to over-summer. If the spawning habitat characteristics of the northern DPS of green sturgeon is any guide, then green sturgeon likely have fewer spawning sites and cumulatively less spawning area than late-fall-run salmon in the Sacramento River. So locating the primary spawning habitats used by green sturgeon in the Sacramento River is essential for assessing the effects of any change in the water temperature regime of the upper Sacramento River.

Because the specific location of green sturgeon spawning and rearing in the Sacramento River is unknown, it is difficult to assess the potential effects of an upstream shift in the water temperature compliance point. Nevertheless, the potential for negative effects highlights the need for more targeted study on green sturgeon larvae and juveniles. To assess the potential effects of a change in the water temperature compliance point on green sturgeon, we recommend that the USBR apply its water temperature model to predict the location of lethal and stressful water temperatures for both the current and the proposed water temperature compliance points. The collection of additional water temperature data in the river would also provide valuable information to assess egg incubation and rearing habitat conditions for green sturgeon.

The water temperature modeling analysis can use metrics produced by recent laboratory studies, which indicate embryo mortality of green sturgeon at temperatures > 68°F (20°C), and potential stress at temperatures > 63–64°F (17–18°C). Optimum temperatures for growth and survival of green sturgeon larvae are believed to range from 59° to 66°F (15° to 19°C) (Mayfield and Cech 2004, as cited in NMFS 2005). Because larval green sturgeon are captured every year at the RBDD RST (RM 243.5) and the GCID screens (RM 206), these two locations can serve as initial sites to track changes in the water temperature regime associated with upstream changes in the compliance point, until future research yields better information about specific rearing habitat locations and conditions. Figure 6-2 demonstrates that mean daily water temperatures rarely exceed 66°F (18.9°C) at RBDD. However, the lack of water temperature data downstream of RBDD makes it difficult to determine if elevated water temperatures pose a risk to larval green sturgeon downstream to, and downstream of, the GCID diversion.

6.6.4 Identify key spawning locations

Green sturgeon appear to prefer deep, fast, turbulent flow over a cobble surface as spawning habitat (Moyle 2002). In the Rogue River, sturgeon have been observed lingering in the vicinity of steep breaks in the channel slope that provide such habitat, though spawning has not been observed directly (D. Erickson, personal communication, July 14, 2005). Historically in the Sacramento River, the majority of this type of habitat likely occurred upstream of the Keswick Dam site (RM 302), however suitable spawning habitat may occur in the canyon reach of the Sacramento River between the confluence of Cottonwood Creek (RM 273.5) and RBDD (RM 243.5), including China Rapids (RM 253.6).

Locating specific green sturgeon spawning locations is essential for conserving and recovering the species, because it could promote a better understanding of habitat preferences that would, in turn, inform efforts to conserve and restore essential habitat. The University of California, Davis has placed several remote logging stations along the Sacramento River (Figure 6-3), so the return of tagged fish in the next few years may provide some insight on the importance of the canyon reach as a destination for green sturgeon spawning. However, one limitation of the current telemetry studies is the long interval between “pings” of the transmitters planted in adult green sturgeon. This feature of the transmitters currently in use helps to preserve battery life, which is important because of the wide-ranging migrations of adult green sturgeon and their infrequent spawning. However, one consequence of the long interval between “pings” is that adult fish can migrate past remote monitoring stations without being detected. Another limitation is the inability to track adult green sturgeon by use of hand-held equipment mounted on a boat, because adult green sturgeon can migrate through the monitored range during an interval between pings. (J. Heublein, personal communication, August 22, 2006).

To facilitate more detailed tracking and identification of specific green sturgeon spawning sites, we recommend that state and federal agencies provide funding to UC Davis to augment the current telemetry studies. Additional studies would involve capturing pre-spawning adults in the Sacramento River and fitting them with acoustic radio tags that can be tracked by hand and boat, to facilitate the identification of specific spawning habitats.

Based on reports of spawning habitat preferences for the northern DPS of green sturgeon, we hypothesize that the vicinity of China Rapids (RM 253.6) may provide suitable spawning habitat for green sturgeon. Though access in this canyon reach is difficult, there are several possible survey methods, including:

- observation from a boat or banks to detect breaching as a courtship behavior (Adams et al. 2002);
- capturing pre-spawning adult and equipping them with radio transmitters that can be tracked using a hand-held radio receiver from a boat to detect the presence and residence time of adults;

- placing additional remote radio receivers more densely in the canyon reach to detect the movement and residence time of tagged adults at a finer spatial scale;
- using a boat-mounted underwater camera to directly observe congregations of adults and possibly spawning activity; and
- direct underwater observation by SCUBA divers.

Any of these survey methods would be applied direct during the peak spawning period from mid-April through mid-June.

6.6.5 Post-spawning holding habitat

Telemetry studies conducted in the Rogue River indicate that adults often hold in deep pools both during their upstream migration and after spawning, and some individuals have been detected using the same pool at different seasons (D. Erickson, personal communication, July 14, 2005). In the Rogue River, these pools are located farther downstream than suspected spawning sites, and they are often associated with deep coves with low velocity flows. Fishing guides have identified reaches of the Sacramento River where adult green sturgeon seem to congregate during early autumn, suggesting that the southern DPS of green sturgeon may also exhibit a post-spawning holding phase. Because post-spawning adults may be holding in a reach of the river during a time when fall-run salmon are migrating upstream, they may be vulnerable to incidental angling pressure. Anecdotal accounts of green sturgeon harvest in the Sacramento River, incidental to the fall-run salmon recreational fishery, suggests that recreational angling may pose a significant threat to the southern DPS (J. Heublein, personal communication, November 14, 2005). Additional studies are needed to determine if the southern DPS of green sturgeon exhibits a post-spawning holding phase and to identify holding habitat, so that the risk of mortality associated with angling can be assessed. Identification of holding habitat in the Sacramento River could also support the definition of geographically explicit angling restrictions to protect green sturgeon while preserving sportfishing in other reaches of the river.

6.6.6 Angling restrictions

As part of the status review for green sturgeon, NMFS determined that the northern DPS of green sturgeon does not require protection. Nevertheless, angling restrictions may be necessary in the Bay-Delta and coastal estuaries in Oregon and Washington to protect the southern DPS, because green sturgeon migrate extensively along the western coast of the United States and likely mingle with the northern DPS in estuaries during summer months. Closure of the commercial harvest in the Columbia River estuary and Willapa Bay in 2001 have likely benefited the southern DPS, but recreational harvest may continue to have an impact. Angling restrictions designed to protect sturgeon will likely cause conflict with sportfishing groups, as evidenced by the public resistance over CDFG's recent attempt in 2005 to limit white sturgeon (*Acipenser transmontanus*) fishing in the Bay-Delta.

In recent years, CDFG game wardens have captured sturgeon poachers in the Delta and lower reaches of Bay-Delta tributaries. Though poachers generally target the more abundant white sturgeon to harvest eggs for caviar, the gear and techniques used to capture sturgeon could also take green sturgeon adults migrating upstream to spawn. The occurrence and severity of poaching mortality is unknown, but the illegal harvest of small numbers of pre-spawning adults likely has significant impacts on the population because of the small number of spawners each year.

6.7 References

Adams, P. B., C. B. Grimes, J. E. Hightower, S. T. Lindley, and M. L. Moser. 2002. Status review for North American green sturgeon, *Acipenser medirostris.* National Marine Fisheries Service, Santa Cruz, California.

Beamesderfer, R. C. 2000. Agenda and notes for green sturgeon workshop, 22-23 March 2000, Weitchpec, California. Oregon Department of Fish and Wildlife, Portland.

Beamesderfer, R. C. P., and M. A. H. Webb. 2002. Green sturgeon status review information. S. P. Cramer and Associates, Gresham, Oregon.

Beamesderfer, R., M. Simpson, G. Kopp, J. Inman, A. Fuller, and D. Demko. 2004. Historical and current information on green sturgeon occurrence in the Sacramento and San Joaquin rivers and tributaries. Prepared by S. P. Cramer & Associates, Oakdale, California for State Water Contractors, Sacramento, California.

CDFG (California Department of Fish and Game). 2002. California Department of Fish and Game comments to NMFS regarding green sturgeon listing. CDFG, Sacramento.

CDFG (California Department of Fish and Game). 2004. Sacramento River winter-run Chinook salmon. Biennial Report 2002-2003. Prepared by CDFG, Habitat Conservation Division, Native Anadromous Fish and Watershed Branch for California Fish and Game Commission.

Cech, J. J. Jr., S. I. Doroshov, G. P. Moberg, B. P. May, R. G. Schaffter, and D. M. Kohlhorst. 2000. Biological assessment of green sturgeon in the Sacramento-San Joaquin watershed (Phase 1). Project No. 98-C-15, Contract No. B-81738. Final report to CALFED Bay-Delta Program.

Clark, G. H. 1929. Sacramento-San Joaquin salmon (*Oncorhynchus tshawytscha*) fishery of California. California Department of Fish and Game Fish Bulletin 17: 20-63.

Deng, X. 2000. Artificial reproduction and early life stages of the green surgeon (*Acipenser medirostris*). Doctoral dissertation. University of California, Davis.

Emmett, R. L., S. L. Stone, S. A. Hinton, and M. E. Monaco. 1991. Distribution and abundance of fishes and invertebrates in west coast estuaries. Volume 2: Species life history summaries. ELMR Report No. 8. NOS/NOAA Strategic Environmental Assessment Division, Rockville, Maryland.

EPIC, CBD, and WaterKeepers (Environmental Protection Information Center, Center for Biological Diversity, and WaterKeepers Northern California). 2001. Petition to list the North American green sturgeon (*Acipenser medirostris*) as an endangered or threatened species under the Endangered Species Act. EPIC, Garberville, California; CBD, Berkeley, California; and WaterKeepers, San Francisco, California.

Erickson, D. L., J. A. North, J. E. Hightower, J. Weber, and L. Lauck. 2002. Movement and habitat use of green sturgeon *Acipenser medirostris* in the Rogue River, Oregon, USA. Journal of Applied Ichthyology 18: 565-569.

Hatton, S. R. 1940. Progress report on the Central Valley fisheries investigations, 1939. California Fish and Game 26: 334-373.

Heublein, J. C. 2006. Migration of green sturgeon Acipenser medirostris in the Sacramento River. Master's thesis. San Francisco State University, San Francisco, California.

Mayfield, R. B., and J. J. Cech, Jr. 2004. Temperature effects on green sturgeon bioenergetics. Transactions of the American Fisheries Society 133: 961-970.

Moyle, P. B. 2002. Inland fishes of California. Revised edition. University of California Press, Berkeley.

Nakamoto, R. J., T. T. Kisanuki, and G. H. Goldsmith. 1995. Age and growth of Klamath River green sturgeon (*Acipenser medirostris*). Project 93-FP-13. U.S. Fish and Wildlife Service, Coastal California Fish and Wildlife Office, Arcata, California.

NMFS (National Marine Fisheries Service). 1993. Biological opinion - Long-term operation of the Federal Central Valley Project and the California State Water Project. Endangered Species Act Section 7 Consultation. NMFS, Southwest Region, Long Beach, California.

NMFS. 2003. Endangered and threatened wildlife and plants: 12-month finding on a petition to list North American green sturgeon as a threatened or endangered species. Federal Register 68: 4433-4441.

NMFS. 2005. Endangered and threatened wildlife and plants: proposed threatened status for Southern Distinct Population Segment of North American green sturgeon. Federal Register 70: 17386-17401.

NMFS. 2006. Endangered and threatened wildlife and plants: threatened status for Southern Distinct Population Segment of North American green sturgeon: final rule. Federal Register 71: 17757-17766.

Tracy, C. 1990. Green sturgeon meeting and comments. Memorandum. Washington Department of Fisheries.

Van Eenennaam, J. P., M. A. H. Webb, X. Deng, S. I. Doroshov, R. B. Mayfield, J. J. Cech Jr., D. C. Hillemeier, and T. E. Willson. 2001. Artificial spawning and larval rearing of Klamath River green sturgeon. Transactions of the American Fisheries Society 130: 159-165.

Van Eenennaam, J. P., J. Linares-Casenave, X. Deng, and S. I. Doroshov. 2005. Effect of incubation temperature on green sturgeon embryos, *Acipenser medirostris.* Environmental Biology of Fishes 72: 145-154.

Yoshiyama, R. M., E. R. Gerstung, F. W. Fisher, and P. B. Moyle. 1996. Historical and present distribution of chinook salmon in the Central Valley drainage of California. Pages 309-362 *in* Sierra Nevada Ecosystem Project: final report to congress. Volume III: Assessments, commissioned reports, and background information. University of California, Center for Water and Wildland Resources, Davis.

Figures

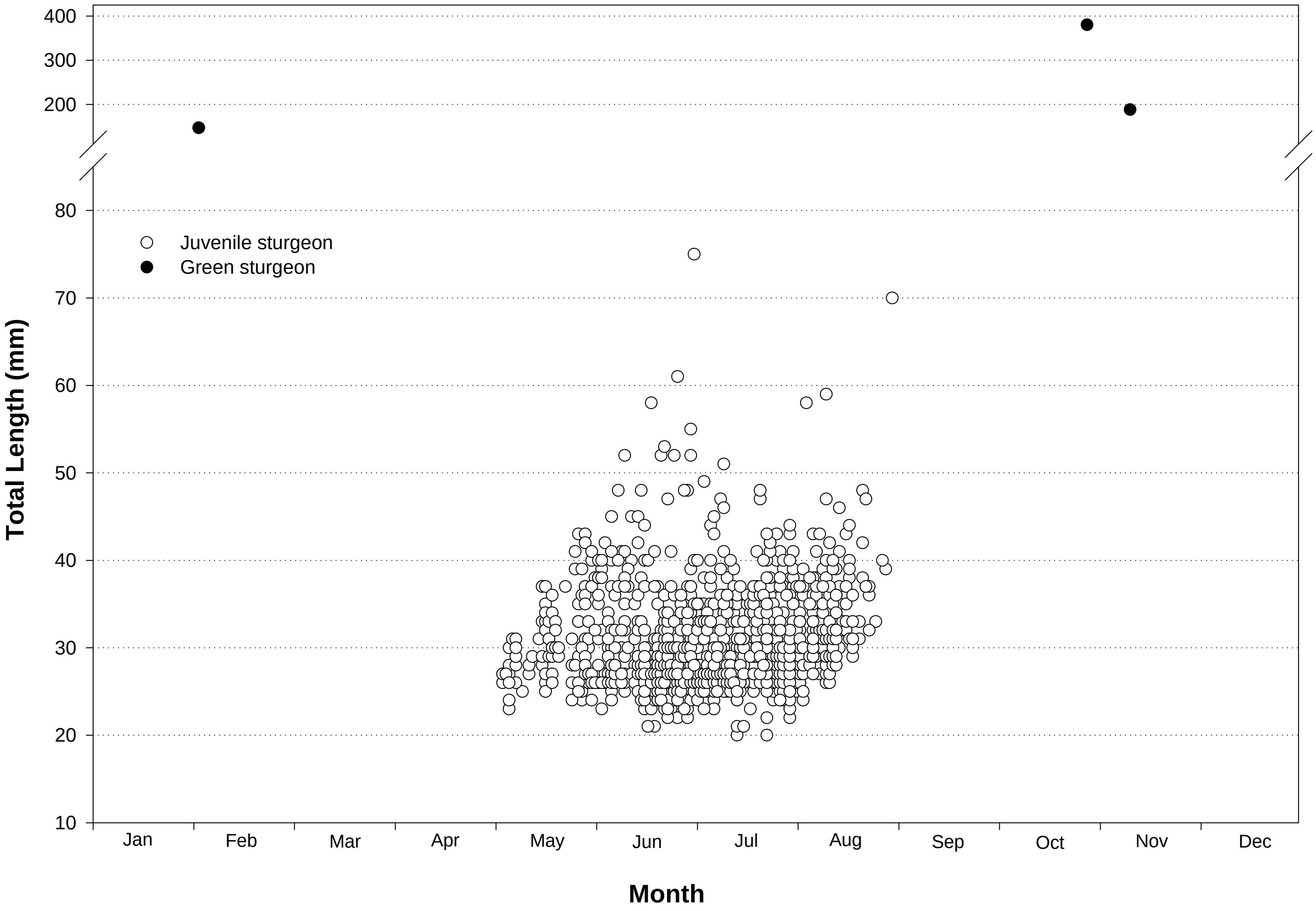


Figure 6-1. Catch of larval green sturgeon at RBDD rotary screw traps (January, 1995 - June 2000, April 2002 - December 2004). The capture of larval green sturgeon at RBDD occurs between May and August. Source: USFWS 2004.

Days of mean water temperature over 66°F at Jellys Ferry gage

	WY															
	1990	1991	1992	1993	1994	1995	1996	1997	1998	1999	2000	2001	2002	2003	2004	2005
Oct	▨ 0	▨ 0	▨ 0	▨ 0	▨ 0	▨ 0	.	.	.	.	.	.	.	.	.	.
Nov	▨ 0	▨ 0	▨ 0	▨ 0	▨ 0	.	.	.	.	.	.	.	.	.	.	.
Dec	▨ 0	▨ 0	▨ 0	▨ 0	▨ 0	▨ 0	.	.	.	.	.	.	.	.	.	.
Jan	▨ 0	▨ 0	▨ 0	▨ 0	.	.	.	.	.	.	.	.	.	.	.	.
Feb	▨ 0	▨ 0	▨ 0	▨ 0	▨ 0	.	.	▨ 0	.	.	.	.	.	.	.	.
Mar	▨ 0	▨ 0	▨ 0	▨ 0	▨ 0	.	.	.	.	.	.	.	.	.	.	.
Apr	▨ 0	▨ 0	▨ 0	▨ 0	▨ 0	.	.	.	.	.	.	.	.	.	4	▨ 0
May	▨ 0	▨ 0	▨ 0	▨ 0	▨ 0	.	.	▨ 0	▨ 0	.	.	.	.	.	.	▨ 0
Jun	▨ 0	▨ 0	▨ 0	▨ 0	▨ 0	.	.	▨ 0	▨ 0	.	.	.	.	.	.	▨ 0
Jul	▨ 0	▨ 0	▨ 0	▨ 0	▨ 0	▨ 0	.	.	▨ 0	.	.	.	▨ 0	.	.	▨ 0
Aug	▨ 0	▨ 0	▨ 0	▨ 0	▨ 0	▨ 0	.	.	.	.	.	.	.	▨ 0	.	▨ 0
Sep	▨ 0	▨ 0	▨ 0	▨ 0	.	.	.	.	.	.	.	.	.	▨ 0	.	▨ 0

Days of mean water temperature over 66°F at Bend Bridge gage

	WY															
	1990	1991	1992	1993	1994	1995	1996	1997	1998	1999	2000	2001	2002	2003	2004	2005
Oct	▨ 0	▨ 0	▨ 0	▨ 0	▨ 0	▨ 0	▨ 0	.	.	.	.	.	.	.	.	▨ 0
Nov	.	▨ 0	▨ 0	▨ 0	.	.	▨ 0	.	.	.	.	.	.	▨ 0	.	.
Dec	.	▨ 0	▨ 0	▨ 0	.	.	▨ 0	.	.	.	.	.	.	▨ 0	.	.
Jan	.	▨ 0	▨ 0	▨ 0	▨ 0	.	▨ 0	.	.	.	.	.	.	.	▨ 0	.
Feb	.	▨ 0	▨ 0	▨ 0	.	.	.	.	.	.	.	.	.	.	.	.
Mar	▨ 0	▨ 0	▨ 0	▨ 0	.	.	.	.	.	▨ 0	.	.	.	.	.	.
Apr	.	▨ 0	▨ 0	▨ 0	.	.	.	.	.	.	.	.	.	▨ 0	.	▨ 0
May	.	▨ 0	▨ 0	▨ 0	▨ 0	.	.	.	.	.	.	.	.	▨ 0	▨ 0	▨ 0
Jun	.	▨ 0	▨ 0	▨ 0	.	.	▨ 0	▨ 0	▨ 0	.	.	.	.	.	.	▨ 0
Jul	▨ 0	▨ 0	▨ 0	▨ 0	▨ 0	.	.	.	▨ 0	.	.	.	.	.	.	▨ 0
Aug	▨ 0	▨ 0	▨ 0	▨ 0	.	.	.	.	.	.	.	.	.	.	.	▨ 0
Sep	.	▨ 0	▨ 0	▨ 0	.	▨ 11	.	.	.	.	.	.	.	▨ 0	.	▨ 0

Days of mean water temperature over 66°F at Red Bluff gage

	WY															
	1990	1991	1992	1993	1994	1995	1996	1997	1998	1999	2000	2001	2002	2003	2004	2005
Oct	▨ 0	▨ 1	▨ 0	▨ 0	▨ 0	▨ 0	.	.	.	▨ 0	▨ 0	.	.	.	.	.
Nov	▨ 18	▨ 0	.	.	▨ 0	.	.	.	.	.	▨ 0	.	.	.	.	.
Dec	▨ 30	.	▨ 0	.	▨ 0	▨ 0	.	.	.	.	.	.	.	.	.	.
Jan	30	.	.	.	▨ 0	.	.	▨ 0	.	.	.	.	.	.	.	.
Feb	.	.	▨ 0	.	.	.	.	▨ 0	.	.	.	.	.	.	.	.
Mar	▨ 0	▨ 0	▨ 0	▨ 0	.	.	.	.	.	.	.	.	.	.	.	.
Apr	.	.	.	.	▨ 0	.	.	.	.	.	.	.	.	.	.	▨ 0
May	.	▨ 0	▨ 0	▨ 0	▨ 0	7	7	▨ 0	.	.	.	.	.	.	.	▨ 0
Jun	.	.	▨ 0	.	16	3	▨ 16	▨ 0	▨ 0	.	.	.	.	.	.	▨ 0
Jul	▨ 0	▨ 0	▨ 0	▨ 0	▨ 6	.	.	▨ 0	▨ 0	.	.	.	.	.	.	▨ 0
Aug	▨ 0	.	.	▨ 0	.	▨ 0	.	.	.	.	.	.	.	.	.	▨ 0
Sep	▨ 0	▨ 0	▨ 0	.	.	.	.	.	.	.	.	.	.	.	.	▨ 0

.	Threshold never exceeded
▨	Data not available for entire month

Figure 6-2. Days when mean daily water temperatures exceed 66°F (18.9 °C) in the upper Sacramento River. Laboratory research suggests that larval green sturgeon require water temperatures below 66°F (18.9 °C). Mean daily water temperatures rarely exceed this temperature threshold in the upper Sacramento River. The lack of water temperature downstream of RBDD makes it difficult to assess if elevated water temperatures downstream of the dam pose a risk to larval green sturgeon.

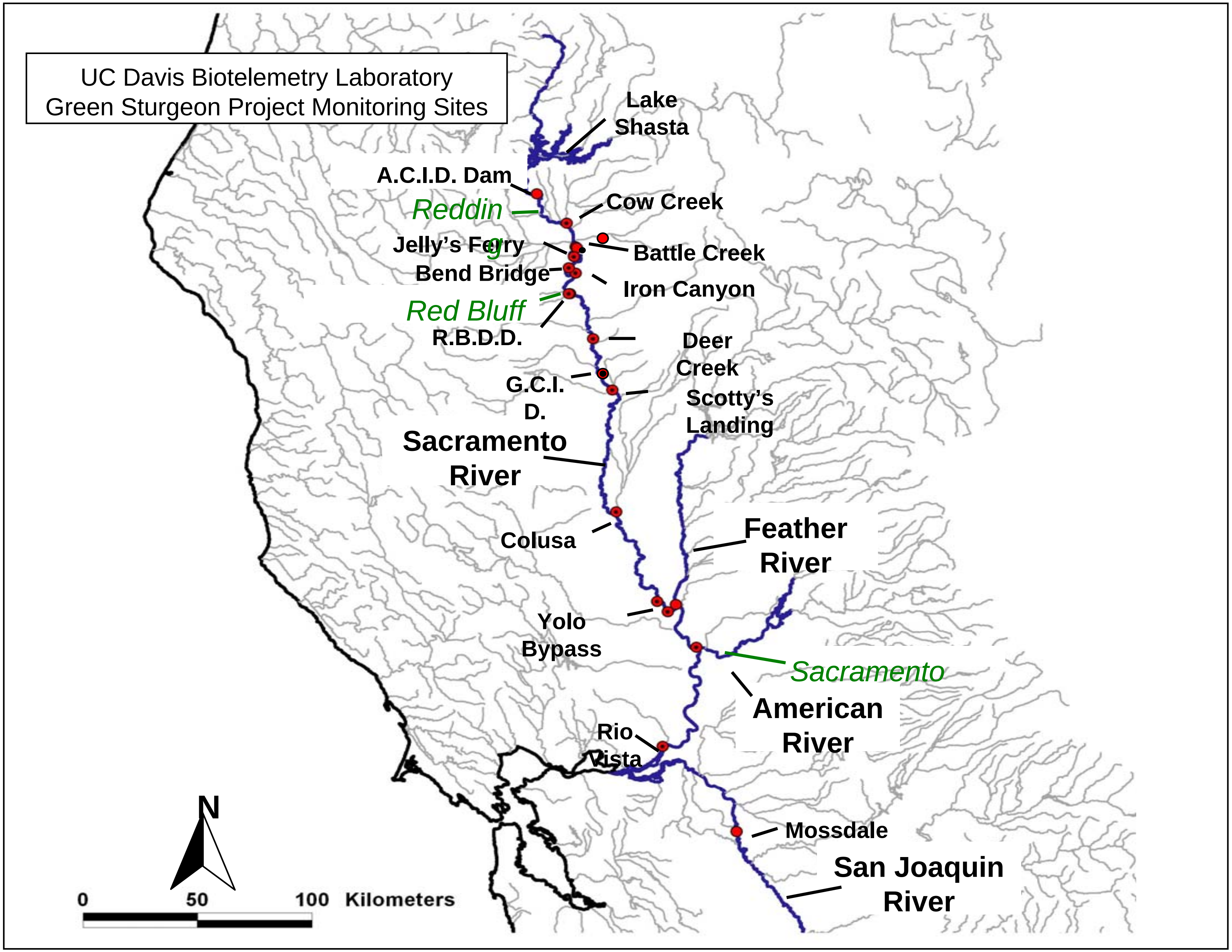


Figure 6-3. General location of UC Davis telemetry monitoring stations in the Central Valley. Source: Heublein 2006

7 BANK SWALLOW

A colonial-breeding migrant from overwintering grounds to the south, the bank swallow (*Riparia riparia*) nests and rears its young in California in spring and early summer, in burrows excavated in steep, freshly eroded river banks that have suitable soils. In response to a sharp decline in the distribution and abundance of bank swallows in the state, the species was listed as threatened under the California Endangered Species Act in 1989. Declines continued through the mid-1990s, and then reversed in a partial recovery toward late-1980s population levels in recent years. Today, over 70% of the state's breeding bank swallows nest along the banks of the Sacramento River and its tributaries. The maintenance of breeding habitat along the Sacramento River in particular will be a key component of any successful bank swallow protection program within the state.

Human activities and modifications of the ecosystem have had significant effects on bank swallow breeding populations and habitat. Indications from recent research, for example, suggest that bank swallows have been affected by conversion of native grasslands to orchards and row crops, which may provide fewer insects on average for foraging (Moffatt et al. 2005). Bank armoring activities have had several immediate and long-term adverse effects on bank swallow populations and habitat including: (1) coverage of steep, fresh surfaces that are suitable for bank swallow nesting, (2) destruction of individual birds (and in extreme cases entire colonies) when construction occurs during breeding season, and (3) localized reductions in the river's ability to create the steep, fresh bank surfaces required by nesting bank swallows. The river's ability to create nesting habitat for bank swallows has also been affected by human modifications to rates and patterns of sediment transport and flow, which together regulate the geomorphic processes that set the rate, type, and timing of bank erosion.

Selection criteria for the focal species considered in this report are provided in Chapter 1. Bank swallows were chosen as a focal species for this study for several reasons:

- the species' threatened status and well-documented decline in the state,
- its heavy reliance on the Sacramento River corridor for breeding habitat in California,
- its sensitivity to how flow, sediment transport, and the lateral migration of the river are managed (with the timing, magnitude, and ramping rates of flow, and decisions about where and when, if ever, to implement bank protection measures being the key adjustable parameters), and
- its sensitivity to management of surrounding landscapes.

7.1 Geographic Distribution, Population Trends and Taxonomy

With a Holarctic breeding distribution and wintering grounds throughout the Southern Hemisphere, the migratory bank swallow is one of the most widely distributed swallows in the world (Garrison 1999). In the Old World, where it is known as the sand martin, the bank swallow breeds throughout Europe and Asia and overwinters in the Mediterranean, Arabia, and Africa (Garrison 1999). In the New World, it breeds in a wide swath from central Alaska south to California and across North America to the eastern coast of the United States and Canada, with wintering grounds in the Caribbean and in Central and South America (Garrison 1999).

7.1.1 General distribution in California

Bank swallow breeding colonies historically occurred throughout California, along large lowland rivers and in coastal areas where alluvial soils exist (CDFG 1992, Garrison 1998). Notably dense populations occurred in coastal southern California, from Santa Barbara County to central San Diego County (Grinnell and Miller 1944, CDFG 1992, Garrison 1998). Today, the southern California populations have been greatly reduced and are thought by many to be completely extirpated from the region (Humphrey and Garrison 1987, Laymon et al. 1988, Garrison 1998). Overall, the bank swallow's range in California has been reduced by an estimated 50% since 1900 (Laymon et al. 1988, CDFG 1997).

A survey in 1987 identified 111 colonies in California and estimated the statewide population of breeding pairs to be 18,800, with roughly 75% occurring along the Sacramento and Feather rivers and their tributaries (Laymon et al. 1988). The Sacramento River population alone was estimated to account for approximately 50% of the state's total in 1987, with distribution from Redding to the Yolo Bypass, in Yolo County. The Feather River population occurs between Oroville and the confluence of the Sacramento and Feather rivers, in Sutter County (Garrison 1998). Since 1988, monitoring in California has focused primarily on the Sacramento River. Relatively little population information exists for the rest of California's Central Valley, but some county-by-county data are available (Garrison 1998). As of 2000, riparian systems of the Sacramento River and its tributaries were estimated to provide suitable breeding habitat for approximately 70% of the bank swallows nesting in California (Hight 2000).

7.1.2 Local distribution

Roughly 75% of the Sacramento River's bank swallow colonies and 85–90% of its burrows and (thus breeding pairs) have been observed annually between Red Bluff and Colusa (RM 243–144) in surveys that began in 1986 (Garrison et al. 1987). Annual surveys are documented in a series of reports by Garrison et al. (1987, 1988), Hight (2000), and Schlorf and (1997, 1998, 1999, 2000, 2001, 2002, 2003, 2004). Surveys of bank swallow populations from Colusa to Verona (RM 143–80) and from Redding to Red Bluff (RM 292–244) have been less frequent, but indicate that bank swallows do occur in those reaches, albeit in lower densities than between Colusa and Red Bluff.

7.1.3 Local population trends

Anecdotal accounts from before 1986 (when surveys began) indicate that the bank swallow was historically common in the Sacramento Valley but was in a protracted period of sharp decline that apparently began in the 1960s (Laymon et al. 1988, CDFG 1997). The coincidence of the sharp bank swallow population decline and the construction of many of the river's bank protection projects is thought to be significant (Remsen 1978; Garrison et al. 1987, 1989; Garrison and McKernan 1994; Schlorff 1997; Garrison 1998). Mechanisms for bank-protection-related declines in bank swallow populations are discussed in Section 7.5. It is conceivable (but difficult to demonstrate) that declines in bank swallow populations began before the 1960's, due to effects of early reclamation and bank protection activities and the construction of Shasta Dam, which, as discussed in greater detail later, may have affected the river's ability to create freshly eroded banks suitable for nesting.

A compilation of annual surveys conducted since 1986 indicates that there has been a general decline in the total number of bank swallow burrows, colonies, and estimated breeding pairs found between Redding and Verona (RM 292–81) (Table 7-1). Peak numbers were observed from 1986 to 1988, with a maximum abundance in 1986, when over 13,000 breeding pairs were present (Figure 7-1, Table 7-1). This was followed by a pronounced decline until 1995, and then a period of somewhat variable but generally low estimated breeding pair totals (ranging from about 5,000 to 7,000) from 1995 through 1998. This was followed in 1999 and 2000 by a general increase (to 8,000–9,000) that has been sustained, with some interannual variability, through 2004. The 2005 survey data, however, indicated a decrease by over a 1,000 breeding pairs from the levels observed from 1999-2004. This decline was due primarily to a decrease in the number of colonies observed in the Butte City to Colusa reach (Tables 7-2 and 7-3). Data for 2006 are not yet available and may be of limited value in tracking overall population trends because the survey was incomplete due to a breakdown of the survey vessel.

Table 7-1. Bank swallow survey data, Sacramento River, RM 292–81 (Redding to Verona).

Year	Total colonies	Total burrows	Average burrow density (burrows/ colony)	Total breeding pairs
1986	72	29,260	410	13,170
1987	66	25,330	380	11,400
1988	76	27,040	360	12,170
1989	62	22,110	360	9,950
1990	54	20,970	390	9,440
1991	47	17,530	370	7,890
1992	57	18,330	320	8,250
1993	49	13,900	280	6,260
1994	42	16,470	390	7,410
1995	47	11,080	240	4,990
1996	52	12,820	250	5,770
1997	52	11,540	220	5,190
1998	42	11,090	260	4,990
1999	57	18,250	320	8,210
2000	46	20,470	450	9,210
2001	51	21,520	420	9,680
2002	57	18,500	320	8,330
2003	61	21,300	350	9,590
2004	56	19,410	350	8,730
2005	52	16,390	320	7,380

The California Department of Fish and Game partitioned the river into 5 reaches in each of their annual surveys: Colusa to Verona (RM 143–81), Butte City to Colusa (RM 168–144), Hamilton City to Butte City (RM 199–169), Red Bluff to Hamilton City (RM 243–200), and Redding to Red Bluff (RM 292–244). Bank swallow nesting has consistently been most productive in the 100 mi (161 km) of river covered by the middle three reaches, from Colusa to Red Bluff (Figure 7-2, Tables 7-2 and 7-3). Breeding population trends were generally similar in each of the three middle reaches; peaks in colonies and burrows (and the estimated number of breeding pairs) occurred between 1986 and 1988, and were followed first by decline, until the mid-1990s, and then partial recovery, from the late 1990s to the present. Close inspection of the population data reveals that the recent partial recovery may have begun in the uppermost reach (from Red Bluff to Hamilton City) first, in 1993, before progressing downstream, first to the middle reach (from

Hamilton City to Butte City) in 1998 and then to the lower reach (from Butte City to Colusa), which had its lowest number of burrows in 1998 and appears to have begun its partial recovery in 1999 (Figure 7-2). Neither the significance nor the cause of the trend in population identified above is well understood. It is possible that the downstream propagating initiation of partial recovery could be due, at least in part, to reach-to-reach differences in geomorphic and anthropogenic constraints on channel migration and/or reach-to-reach differences in the effects of flow regulation. Further investigation of the differences among reaches in habitat conditions and physical processes may be important in understanding the mechanisms driving observed trends in population dynamics on the river. For example, the Butte City to Colusa reach was the only reach showing a pronounced decline in colonies (Table 7-2) and breeding pairs (Table 7-3) in 2005, but at present we don't know why this localized decline occurred. More specific discussion of the origins and significance of observed population trends are discussed below in Section 7.5, with particular focus on metapopulation dynamics in Section 7.5.5.

Table 7-2. Number of bank swallow colonies by reach, Sacramento River, 1986-2005.

Year	Reach					
	RM 143–81 Colusa to Verona*	**RM 168–144 Butte City to Colusa***	**RM 199–169 Hamilton City to Butte City**	**RM 243–200 Red Bluff to Hamilton City***	**RM 292–243 Redding to Red Bluff***	**Totals for RM 292–81**
1986	13	15	15	23	6	72
1987	12	13	16	20	5	66
1988	_9*_	18	28	_16*_	_5*_	76
1989	6	_14*_	21	_16*_	_5*_	62
1990	6	15	15	15	3	54
1991	6	9	14	13	_5*_	47
1992	9	14	15	14	_5*_	57
1993	8	15	11	10	_5*_	49
1994	6	11	10	10	_5*_	42
1995	4	12	11	15	5	47
1996	5	12	11	19	_5*_	52
1997	7	14	14	12	_5*_	52
1998	0	7	12	18	_5*_	42
1999	5	12	13	22	_5*_	57
2000	8	8	11	14	_5*_	46
2001	_8*_	9	14	15	_5*_	51
2002	_8*_	17	14	13	_5*_	57
2003	_8*_	11	20	17	_5*_	61
2004	_8*_	13	15	15	_5*_	56
2005	_8*_	6	16	17	_5*_	52

*Some reaches were not surveyed in every year. In such cases, CDFG estimated the likely number of colonies in that reach based on data from other years. These estimates for non-surveyed reaches are italicized and marked with an asterisk.

Table 7-3. Estimated number of bank swallow breeding pairs by reach on the Sacramento River for 1986–2005.

Year	Reach					
	RM 143–81 Colusa to Verona*	RM 168–144 Butte City to Colusa	RM 199–169 Hamilton City to Butte City	RM 243–200 Red Bluff to Hamilton City	RM 292–243 Redding to Red Bluff*	Totals for RM 292–81**
1986	1,120	2,730	3,390	5,190	750	13,170
1987	1,670	2,970	2,280	3,840	560	11,400
1988	840	3,510	4,310	2,930	_580*_	12,170
1989	340	2,960	3,140	2,930	_580*_	9,950
1990	440	3,350	2,180	3,100	370	9,440
1991	840	2,750	1,780	1,940	_580*_	7,890
1992	740	3,080	2,030	1,820	_580*_	8,250
1993	720	2,350	880	1,720	_580*_	6,260
1994	1,110	2,190	1,530	2,000	_580*_	7,410
1995	240	940	1,130	2,100	580	4,990
1996	320	1,210	1,120	2,540	_580*_	5,770
1997	330	970	1,170	2,150	_580*_	5,200
1998	0	810	1,370	2,220	_580*_	4,990
1999	170	1,130	2,910	3,420	_580*_	8,210
2000	480	1,260	3,120	3,780	_580*_	9,210
2001	_480*_	2,100	3,590	2,940	_580*_	9,690
2002	_480*_	1,360	2,550	3,370	_580*_	8,340
2003	_480*_	1,810	3,000	3,730	_580*_	9,600
2004	_480*_	1,770	2,800	3,110	_580*_	8,740
2005	480*	960	2,310	3,050	590*	7,380

*Some reaches were not surveyed in every year. In such cases, CDFG estimates the likely number of burrows and breeding pairs in that reach based on data from other years. These estimates for non-surveyed reaches are italicized and marked with an asterisk.

Estimated number of breeding pairs = total observed burrows 0.45, rounded to the nearest 10, based on data and methods developed by Garrison et al. (1987)

**Totals for the full survey area (RM 292-81) may not match the sum of the pairs for each reach due to rounding error

7.1.4 Taxonomy

The bank swallow is a member of the family Hirundinidae and is the smallest of the nine North American swallow species (Sibley 2000). Eight subspecies of bank swallow were listed by J. L. Peters (Mayr and Greenway 1960), but only 3–4 of them appear to be recognized by species experts (Garrison 1999). North America supports *Riparia riparia riparia*, which breeds throughout the continent, and also *R. r. diluta*, a slightly paler and grayer vagrant of the arctic (Garrison 1999). Intermediates between *R. r. diluta* and *R. r. riparia* occur where their ranges overlap (Garrison 1999). Two races of *R. r. riparia* (i.e., *R. r. maximiliani* and *R. r. ijimae*) have been proposed based on differences in size (Arny 1952, Oberholser 1974, both as cited in Garrison 1999) and other diagnostic factors (Phillips 1986, as cited in Garrison 1999), however, most experts do not currently recognize the validity of these two races (B. Garrison, pers. comm.., 2006).

7.2 Life History and Habitat Requirements

The bank swallow is a highly social migratory land bird which typically breeds in colonies of 10 to as many as 2,000 active nests (Garrison 1999). Isolated individual nests occur on rare occasion (Cramp 1988, Hoogland and Sherman 1976, Turner and Rose 1989), and there have been reports of a few large colonies with up to 3,000 active nests (Garrison 1998). Nests are built in burrows which the birds excavate in steeply sloped banks with friable soils (Garrison 1998, 1999). Most of California's hundred or so colonies occur in steep river banks and bluffs, in the riparian ecosystems of large lowland rivers of the northern half of the state. A few colonies still persist in coastal bluff habitats in California. Bank swallows nesting in riverine systems generally benefit from bank erosion caused by higher winter stream flows, which renews nesting habitat while they are away in overwintering habitats to the south. Throughout California, much of the breeding habitat of the bank swallow has been lost or threatened by flood control and bank protection projects (Schlorff 1997, Garrison 1998, Moffatt et al. 2005).

7.2.1 Life history

Bank swallow arrivals in California begin in mid-March or early April (Humphrey and Garrison 1987, Laymon et al. 1988, Garrison 1998) (Table 7-4). Nesting colonies are established in nearly vertical eroding banks. The bank swallow breeds and usually lays a clutch of 4–5 eggs in April (Garrison 1998, 1999). Young typically hatch in May after 14–16 days of incubation, and 2–3 young are fledged after another 18–24 days in the nest—typically by June or early July (Ehrlich et al. 1988, Garrison 1999). In California, bank swallows typically make only one nesting attempt per year, although swallows in one colony were observed re-nesting after a nest failure in late May due to partial bank collapse (B. Garrison, pers. comm., 2005).

Table 7-4. Generalized timing of bank swallow life history stages for birds breeding in Sacramento Valley, California.

Life stage	Month																							
	Jan		Feb		Mar		Apr		May		Jun		Jul		Aug		Sept		Oct		Nov		Dec	
Winter range	■	■	■	■	■	▒												■	■	■	■	■	■	■
Migration to breeding range				▒	■	■	■	▒																
Pair formation					▒	▒	■	■	■	▒														
Egg incubation								▒	■	■	■													
Nestling rearing									▒	■	■	■	▒	▒										
Fledgling/juvenile rearing												■	■	■	■	▒								
Migration to winter range													▒	■	■	■	■							

▒	Period of off peak activity
■	Period of peak activity

After young fledge and until fall migration begins, juvenile and adult bank swallows roost together in trees and shrubs and on exposed roots, banks, and woody debris, and on sand and gravel bars (Petersen 1955, Cramp 1988, both as cited in Garrison 1998). After breeding, flocks generally remain close to nesting sites until all young have fledged (Garrison 1998). Juveniles disperse from colonies before adults, generally starting in late June and early July. Breeding areas in California become essentially devoid of bank swallows by mid-July to early August (Garrison 1998). Stragglers have been recorded in southern California as late as early November, but the species is essentially absent from the state during the winter period (Small 1994).

7.2.2 Habitat requirements

7.2.2.1 Nesting habitat

Nesting sites in California are usually immediately adjacent to still or running water. Colonies are most commonly located along natural river banks, but occasionally occur at gravel extraction sites. Larger rivers, streams, and lakes (and some gravel extraction sites) provide large, open areas for flying around nest burrows (Hjertaas 1984). Nesting sites must be renewed regularly, when swallows are absent, by an erosive process such as lateral river migration (Garrison 1999).

Nests are built in subterranean burrows in nearly vertical banks. Hence soils must be friable enough for excavation, and at the same time coherent enough that they do not collapse. Burrows generally run parallel to the ground surface and perpendicular to the bank face and are typically 2–3 ft (0.6–0.9 m) deep. Nest cavities are located at the ends of burrows and are enlarged upward and to the sides (relative to the burrow corridor), with nest cavity floors remaining level with burrow floors (Hickman 1979). Burrows are generally located in the upper portions of banks or bluffs, with burrow density decreasing from top to bottom (Sieber 1980). Along the Sacramento River, burrows are generally located in the upper third of occupied banks (B. Garrison and R. Schlorff, pers. comm., 2005).

Ideal banks or bluffs are a minimum of 7 ft (2 m) tall, and more often > 10 ft (3 m) in height, providing protection against predation and inundation by rising flows (Garrison 1998). In one survey of 32 nesting colonies in California, the average height (above water level at the time of survey) and length of colonized banks were 10.8 ft (3.3 m) (standard deviation [SD]=1.7, range=1.3–7.3, n=32) and 1,493 ft (455 m) (SD=441, range=13–1900, n=32), respectively (Humphrey and Garrison 1987). Although there is no empirical evidence that larger colonies have greater nesting success, larger colonies tend to be reoccupied more consistently in successive years than smaller colonies (B. Garrison, unpublished data, as cited in Garrison 1998).

Vegetation associated with bank swallow colony sites varies; breeding sites are typically selected by suitability of the nesting bank, not local vegetation (Garrison 1998, 1999). Colonies can occur immediately below irrigated row crops and orchards as well as under banks that are covered by natural vegetation (Garrison 1998), but seldom are colonies located in banks of eroding riparian forests which have roots interspersed throughout the sediment (G. Golet, personal communication, 2006). Vegetation is usually absent in the immediate vicinity of bank swallow nest sites—particularly around individual nest burrows.

The territory of any given swallow is generally centered on its burrow, which is often no more than 8 in (20 cm) from its nearest neighbor (Petersen 1955, as cited in Zeiner et al. 1990). Humphrey and Garrison (1987) report a mean distance between burrows of 5 in (13 cm) (SD=1.1, range=1–59) for Sacramento River colonies. The area immediately around the nesting burrow is defended early in nesting period, while only the burrow itself is defended after eggs hatch (Garrison 1998). Nest owners attack birds that try to build nests within the immediate vicinity (i.e., within a 3–5 in [8–12 cm] radius). The close spacing of burrow entrances relative to preferred burrow depth inevitably leads to occasional joining of new and existing tunnels, typically resulting in abandonment of one or both of the two nesting attempts (Garrison 1998). Males typically abandon burrows that do not attract mates, excavating new burrows, and thus leading to substantial burrow surpluses within colonies (Kuhnen 1985).

At colony sites that are reoccupied in successive years, bank swallows generally dig new burrows—particularly in banks that have collapsed from erosion or anthropogenic activities (Petersen 1955, Hickman 1979, Cramp 1988). Some old burrows are reused, particularly if they can be enlarged and deepened. Excavation activities such as these are thought to be part of the pair bonding process (Petersen 1955, Hickman 1979). In burrows that are reoccupied, nests are typically removed and replaced with new nest material (Petersen 1955). Males that produce a second brood within a breeding season tend to reuse their first-brood burrows more often than females (Sieber 1980).

7.2.2.2 Foraging habitat

Bank swallow colonies are generally located in areas with sufficient food resources nearby (i.e., near foraging habitats that support large amounts of insect biomass) (Garrison 1998). Adult bank swallows do not generally range very far from their nest burrows during the breeding season (Grinnell and Miller 1944, as cited in Zeiner et al. 1990). Foraging is generally focused within 660 ft (200 m) of the colony when young are being fed (Garrison 1999, Moffatt et al. 2005). Bank swallows travel farther away (up to 5–6 mi [8–10 km]) if good foraging is not available nearby (Mead 1979, as cited in Garrison 1989). Bank swallows forage from dawn to dusk, feeding on flying terrestrial and aquatic insects that are captured almost exclusively on the wing over lakes, ponds, rivers and streams, meadows, fields, pastures, bogs, and occasionally over forests and woodlands (Stoner 1936, Gross 1942, Turner and Rose 1989, Garrison 1989). Foraging height above the water or ground averages about 15 ft (5 m) and ranges from just above the surface to as high as 33 ft (10 m) (Garrison 1998). Mayflies and other aquatic insects can be caught a few inches above water as they emerge from their immature stages (CDFG 1992). Ground feeding occurs only occasionally, usually in instances when suitable insects are heavily concentrated in localized areas on the ground (Clegg 1977, Hobson and Sealy 1987). Bank swallows may feed singly, in pairs, or in flocks. Flock feeding occurs most frequently when there is a highly localized source of prey (Stoner 1936, Turner and Rose 1989). In one recent analysis it was shown that extinction rates of colonies decrease with increasing proximity to grasslands (Moffatt et al. 2005), presumably due to high insect abundance associated with grassland habitats (Humphrey and Garrison 1987, Drake and Farrow 1988).

7.2.3 Biotic interactions

Bank swallows are extremely social, gathering with other individuals on wires and roots when they are away from their nests. Preening birds are often observed perched together closely with shoulders touching (Garrison 1998).

7.2.3.1 Coloniality

Coloniality in bank swallows may impart improved foraging efficiency and prey evasion, with colonies acting as information centers for finding food and reducing predation on eggs and nestlings via "selfish-herd" and group mobbing behaviors (Ehrlich et al. 1988, Brown and Brown 1996). Observations of cliff swallows provide instructive cases in point, assuming that bank swallows and cliff swallows exhibit similar group behaviors when foraging and evading prey. For example, group-related improvements in food-finding for cliff swallows are evident from observations of individuals that return to their colonies after unsuccessful foraging attempts and follow successful foragers back to proven food sources (Ehrlich et al. 1988).

There are several indications, however, that foraging-related benefits of colonial living do not increase with increasing colony size. For example, ten-day-old cliff swallow nestlings in large colonies have been shown to weigh less than their counterparts in small colonies (Ehrlich et al. 1988). Moreover, in times of food shortage, survival rates of young appear to increase with decreasing colony size (Ehrlich et al. 1988).

In contrast, group mobbing, which helps drive away predators (Ehrlich et al. 1988), is expected to increase in overall effectiveness with increasing colony size, because bigger colonies should generally produce bigger (and presumably more effective) mobs. Adverse effects of predator swamping (i.e., with larger colonies attracting higher concentrations of predators and thus leading to increased predation rates) would generally work to undermine benefits of group mobbing. In one study it was observed that, within a given colony, isolated burrows produced fewer young per burrow than closely spaced burrows (Freer 1977, as cited in Garrison 1989), consistent with burrows at the less populated edges of colonies being more vulnerable to predation, and thus alternatively supporting the predator-mobbing and predator-swamping hypotheses for relationships between predators and bank swallows.

If there is a net group-related benefit of predator evasion (i.e., despite effects of predator swamping), and it increases with increasing colony size faster than group-related benefits of food-finding are reduced, then there may be a net advantage of increased colony size. This hypothesis is consistent with the observation that the risk of extinction of bank swallow colonies on the Sacramento River appears to decrease significantly with increasing colony size (Moffatt et al. 2005). Decreased risk of extinction for larger colonies might also be explained by nesting habitat conditions if larger colonies tend to occur on larger banks which are more likely to persist from year to year, while smaller colonies may occur on smaller banks that are located in more dynamic locations. Additionally, larger colonies may be less likely to go extinct simply because there more individuals per colony so that, if all else remains equal, there is a greater likelihood of at least some birds returning to the site in subsequent years.

7.2.3.2 Predation

American kestrels (*Falco sparverius*) and peregrine falcons (*F. peregrinus*) can take bank swallows on the wing by chasing individuals that have been singled out from mobbing flocks (Garrison 1998, 1999). Aerial captures of bank swallows by raptors are probably most common during fledging periods, when relatively vulnerable young are abundant (Garrison 1999). Ravens (*Corvus corax*) are another potential predator on both adults and young (Petersen 2005), although they are rare in the Sacramento Valley and unlikely to be an important predator along the middle Sacramento River (Garrison, personal communication, 2006). Herons, egrets, snakes, rats, raccoons and skunks prey on eggs, juveniles, and adults inside burrows (Zeiner et al. 1990, Garrison 1998), with snakes appearing to be the most important nest predator (Garrison 1998).

The main predators of bank swallows during their annual migrations and in their wintering range have not been systematically identified (Garrison 1998). The overall risk of predation for a given colony is likely to be a function of local predator populations, bank height, bank slope, and amount of vegetation—especially roots that might facilitate climbing by terrestrial predators and provide perches for aerial predators. Suitability criteria for bank height and bank slope as they relate to predator evasion are discussed in greater detail in Section 7.2.4.

7.2.3.3 Other biotic interactions

Ectoparasites may reduce the reproductive success of bank swallows (Szep and Moller 1999). Reuse of old nests is probably avoided due to increased likelihood of infestation by fleas (*Ceratophyllus* spp.) and other ectoparasites in nests (Haas et al. 1980, Garrison 1999). This implies a nearly continuous need for new nests, and has important implications for bank swallows on the Sacramento River, where fresh surfaces for burrow construction are generated by lateral channel migration and the associated process of bank erosion.

During fall migration, bank swallows occur in mixed-species flocks with barn swallows (*Hirundo rustica*), cliff swallows (*H. pyrrhonota*), and tree swallows (*Tachycineta bicolor*) (Garrison 1998). Interspecific competition has not been widely documented, but there are records of European starlings (*Sturnus vulgaris*) and house sparrows (*Passer domesticus*) usurping nest burrows from bank swallows (Garrison 1998). Characteristics of suitable nest sites for bank swallows and northern rough-winged swallows (*Stelgidopteryx serripennis*) overlap; both species build burrows in vertical banks in friable substrates, and sometimes occupy banks in close proximity to one another, but the extent of competition (if any) for burrow sites is unknown (Garrison 1998, 1999). Other birds known to nest in bank swallow colonies include barn owl (*Tyto alba*) and belted kingfisher (C*eryle alcyon*) (Garrison 1999) and cliff swallows (G. Golet, personal communication, 2006). Brood parasitism by brown-headed cowbirds is very rare (Ehrlich et al. 1988, Garrison 1999). Compared to other small-bodied land birds, bank swallows appear to be minimally affected by exotic avian species (Garrison 1998).

Speakman et al. (2000) hypothesized that nocturnal foraging behavior of insectivorous bats in northern Scandinavia might have evolved, at least in part, due to competition with bank swallows for aerial insects, although they found more support for alternative hypotheses. Cliff swallows (*Petrochelidon* [=*Hirundo*] *pyrrhonota*) nesting in large colonies on bridges or cliffs along the river might be a potential competitor for aerial insects, but this seems unlikely except at a very localized level, particularly since most cliff swallows along the Sacramento River nest under bridges that are not in close proximity to bank swallow colonies (B. Garrison, personal communication, 2006). The uncertainties in interspecific relationships of bank swallows are worthy of further academic attention, but are not critical concerns for the Sacramento Ecological Flows Study, because they do not bear directly on potential management actions.

7.2.4 Habitat suitability relationships

Using the general methodology developed by USFWS for habitat evaluation (USFWS 1980a), Garrison (1989) developed a habitat suitability index (HSI) model for breeding bank swallows for application to their entire breeding range in the continental United States. Garrison's (1989) HSI model is, in essence, a series of hypotheses about species-habitat relationships that can be used to evaluate bank swallow nesting habitat suitability as a function of soil texture, bank slope, bank height, and bank length (Figure 7-3).

7.2.4.1 Soil texture

For a bank to be suitable for nesting it must contain one or more exposures of soils with textures that are both amenable to excavation and at the same time coherent enough to maintain stable burrows. Suitable exposures must also have sufficient thickness (i.e., ≥ 0.8 ft [0.25 m]) and depth perpendicular to the bank (order 2–3 ft [0.67–1 m] or more) to accommodate typical burrow dimensions. Suitable textures include sand, loamy sand, and sandy loam with clay content less than 18%, medium loam with clay content less than 18–25%, and silty loam with clay content less than 20–25% and sand content greater than 25–30% (Figure 7-3A). In the HSI model, if at least one exposure of soil with a suitable texture, thickness, and depth is present, the suitability index (SI) for variable number one (i.e., soil texture) is equal to 1. For all other exposures, the SI for soil texture is equal to 0.

7.2.4.2 Bank slope

Bank swallows require steep banks for protection from predators and inclement weather (Garrison 1989). A case in point for this requirement is the increased predation by black rat snakes and eventual colony abandonment observed by Blem (1979 as cited in Garrison 1989) as bank steepness at one colony decreased over time. Garrison's (1989) review of bank slope data, which included observations from the Sacramento River (Garrison et al. 1987) and elsewhere, led him to conclude that the SI for bank slopes should be 0 (unsuitable) for slopes less than 60 degrees and 1.0 (optimal) for slopes greater than 80 degrees .The relationship between suitability and bank slope (HSI variable number two) from Garrison's (1989) synthesis is the solid line plotted in Figure 7-3B. The dashed line in Figure 7-3B is slightly different, with constraints based solely on Sacramento River data—a subset of the observations used by Garrison (1989) to construct the relationship shown by the solid line. Bank slopes of nesting colonies on the Sacramento River range from 63.3–96.7 degrees (Garrison et al. 1987), implying that the local threshold for suitability is about 63 degrees. Optimal bank slope (above which SI=1.0) for the Sacramento River case (dashed line in Figure 7-3B) is 81.5 degrees, somewhat arbitrarily calculated as the mean (83.3 degrees) minus 2 times the standard error (0.9 degrees). The differences between the river-specific and overall relationships of Figure 7-3B are small enough that they can be ignored in most practical applications, but they are noted here in any case for completeness.

7.2.4.3 Bank height

The third HSI variable, bank height, is, like bank slope, largely related the need for protection from predation. Swallows that nest in higher bank positions presumably have less risk of predation from small terrestrial mammals (Hoogland and Sherman 1976 and Hickman 1979, both as cited in Garrison 1989). Some predators, such as snakes, squirrels, large mammals, egrets, herons, and raptorial birds, may still be able to gain access to nesting colonies on higher banks (Garrison 1989), but it seems likely that higher banks nevertheless generally impart at least some protection from most predators. Bank swallows on the Sacramento River have been observed to concentrate nests in the upper third of suitable banks (B. Garrison and R. Schlorff, pers. comm., 2005), consistent with higher bank positions being more favorable. Garrison's (1989) relationship between SI and bank height is the solid line shown in Figure 7-3C. As in Figure 7-3B (discussed above), the dashed line in Figure 7-3C is slightly different, because it is based on site-specific data rather than a broad synthesis of widely scattered data from the entire continental United States. Garrison et al. (1987) found that bank heights at breeding colonies on the Sacramento River ranged from 4.3 to 24.3 ft (1.3–7.3 m), implying that the minimum bank height threshold is approximately 4 ft (1.3 m)—with lower heights having SI equal to 0 (Figure 7-3C).

Optimal bank height (above which SI=1.0) is somewhat arbitrarily calculated to be 2.7, equal to the mean 10.8 ft (3.3 m) minus 2 times the standard error 1.0 ft (0.3 m) reported by Garrison et al. (1987). Measurements of the height above base-flow water surface during the nesting season for individual burrows (as opposed to the colony as a whole) are not currently available but would provide important additional constraints on the SI of bank height.

7.2.4.4 Bank length

The fourth habitat variable that Garrison (1989) incorporated into his HSI model is total length of suitable bank (Figure 7-3D). This accounts for the constraint that suitable exposures must provide habitat for a colony of many nests and reflects the need for continuous strata (or large lenses) of suitable material. The minimum bank length necessary for a nesting colony has not been quantified, but data from the Sacramento River show that larger colonies are found on longer banks (Garrison 1989). In a study of 32 colonies on the Sacramento River (Garrison et al. 1987), bank lengths were found to range from 43 to 6,233 ft (13 to 1,900 m). In the HSI model (Garrison 1989) banks greater than 66 ft (20 m) in length are considered to be optimal (with SI=1) and banks with zero length have SI=0 (solid line in Figure 7-3C). Data specific to the Sacramento River suggest that the minimum and optimal bank length thresholds in the HSI could be revised to 43 and 131 ft (13 and 40 m) respectively (dashed line in Figure 7-3D), so that they reflect the observed minimum and a maximum that is based on the length of colonies within a bank rather than total bank length (see Garrison et al. 1987 for more details).

7.2.4.5 Overall suitability

Overall habitat suitability for bank swallow nesting in the HSI model is calculated from (Garrison 1989):

$$\mathrm{HSI}=\mathrm{SI}_1 \cdot (\mathrm{SI}_2 \cdot \mathrm{SI}_3 \cdot \mathrm{SI}_4)^{1/3}$$

where subscripts indicate variable number. The HSI equation assumes that habitat suitability is coupled especially tightly to soil texture (SI_1) and modified by bank slope (SI_2), height (SI_3), and length (SI_4). Note that if the suitability index of any of the four variables equals zero, the total habitat suitability index will also equal zero. Trends in habitat quality and quantity at each location can be quantified by tracking fluctuations in the number of "habitat units", calculated as the product of HSI and the total area (i.e., bank length · bank height) of available nesting habitat (Garrison 1989).

7.2.4.6 Proposed addenda to the HSI model

Review of the literature and bank swallow habitat data from the Sacramento River colonies suggest several additional variables that might be considered in any revision of the existing HSI model, particularly to tailor it for specific application along the Sacramento River. These potential additional parameters are discussed below. However, more intensive study is needed before a new or revised HSI model can be developed since even minor changes in the existing SI functions, let alone addition of new SI variables, may lead to unreasonable changes in overall habitat suitability indicated by the HSI model (B. Garrison, pers. comm.. 2006).

Distance to grassland

One assumption required in the HSI model described above is that availability of food is not a limiting factor (Garrison 1989). This is supported for most situations by indications throughout the literature that bank swallows can adaptively feed on a variety of insect prey in a variety of foraging habitats (e.g., see summaries in Garrison 1998). However, in one recent study of Sacramento River bank swallows (Moffatt et al. 2005), the probability of colony extinction was shown to be related to the distance to the nearest grassland, with distances greater than 660 ft (200 m) corresponding to higher extinction risks—presumably because grasslands provide optimal foraging habitat. Hence, for specific application of the HSI to the Sacramento River, it seems appropriate to consider adding a suitability index relationship (such as the one shown in Figure 7-4A) for the distance to the nearest grassland.

Review of the literature and discussions with local bank swallow experts (B. Garrison and R. Schlorff, pers. comm., 2005) support consideration of three additional habitat suitability relationships (Figure 7-4B–D) to account for the general degradation in habitat quality that occurs when bank exposures are not regularly renewed by erosion and the effects of high flow events during the nesting season.

Rate of bank erosion

There are several mechanisms of habitat degradation over time if bank surfaces are not renewed by erosion. For example, minor sloughing of bank surfaces can reduce bank slope and create piles of debris below burrows, increasing susceptibility of nests to predation. Vegetation encroachment over time may further promote nest predation by providing climbing holds and perching points for predators. Finally, it is thought that detrimentally high populations of ectoparasites (e.g., fleas and lice) may build up in nests over time, leading to abandonment of colonies that are not renewed by erosion. Most of the colonies in the Sacramento valley are used for no more than 7 consecutive years in the absence of erosion (Moffatt et al. 2005). Observations of colonies on the river suggest that suitability of banks remains high (SI=1) for three years, and then drops rapidly, with banks becoming unsuitable (SI=0) after 5–7 years (B. Garrison, pers. comm., 2005), implying a suitability index relationship similar to the one shown in Figure 7-4B.

Flows

High flows during nesting season are generally infrequent on the Sacramento River but nevertheless have the potential to adversely affect bank swallow colonies. Although there is general disagreement on the exact magnitude of flow required to initiate substantial bank erosion (see Chapter 3), there is growing evidence that flows in the 20,000–25,000 cfs range can begin to erode some banks, causing partial bank collapse that can result in nest failure if swallows are present (see Section 7.5.2 for more details). Flows above 50,000–60,000 cfs are almost certain to cause widespread (i.e., spatially extensive) bank erosion, leading to partial or complete colony failure at many sites if breeding bank swallows are present. A proposed suitability index relationship for high flows during the nesting season is shown in Figure 7-4C. A habitat suitability index relationship for high flows during the fall or winter is not shown in Figure 7.4 but might follow the inverse of the function shown in Figure 7-4C, with SI increasing from 0 at 20,000 cfs to 1 at 50,000 cfs, reflecting the bank swallow's need for fresh surfaces for nesting habitat (see Section 7.4 and 7.5 for further discussion). The inference that suitability can either increase or decrease with flow depending on timing is consistent with the otherwise seemingly contradictory observation (Moffatt et al. 2005) that rates of colonization and extinction both increase with increasing discharge.

Stage

High flows that cause large increases in stage (i.e., water surface elevation) above baseflow levels during nesting season may inundate nests and cause direct mortality of bank swallows. A proposed suitability index relationship for increases in stage during the nesting season is shown in Figure 7-4D. It reflects observations (from anecdotal accounts and data from Garrison et al. 1987) that a few nests would be affected by stage increases of 3.3 ft (1 m), that considerably more would affected by an increase in stage of 6.6 ft (2 m), and that a majority of nests along the Sacramento River would be inundated by an increase in stage of 9.8 ft (3 m) or more (as indicated by the observed tendency of burrows to be located within the upper third of banks that have a mean total height of roughly 10.8 ft [3.3 m]).

Garrison's (1989) HSI model is useful for focused field assessment of the physical characteristics of banks as they apply to suitability for bank swallow nesting (Figure 7-3). The habitat suitability relationship for distance to nearest grassland (Figure 7-4A) attempts to factor in the importance of landscape context as it relates to potential food limitation, while the other three new SI relationships of Figure 7-4 address the potential importance of flow conditions (whether natural or managed) and focus on particular mechanisms of nest failure and bank swallow mortality due to high flows during the nesting season. The new relationships (Figure 7-4), like those in Garrison's (1989) original model (Figure 7-3), are in essence a series of hypotheses about species-habitat relationships which are based on reasonable assumptions that can be tested with focused studies (see Section 7.7 for further discussion of key hypotheses, uncertainties, and potential actions to address them).

7.3 Conceptual Model of Historical Conditions in the Sacramento Valley

Bank swallows nesting along the Sacramento and other lowland alluvial rivers have adapted to breeding in a naturally dynamic system. Most colony sites are suitable for only a few years before they are either eroded away, become too accessible to predators due to minor bank sloughing or vegetation encroachment, or build up detrimentally high levels of ectoparasites (forcing abandonment). Under historical conditions, progressive meander migration and cut-off processes created a shifting mosaic of freshly eroded banks along the river corridor. The river and its banks were located within a larger shifting mosaic of riparian habitats, winding across a broad alluvial floodplain, and thus providing bank swallows with access to a variety of potential foraging habitats, including grassland and herbaceous vegetation, off-channel wetlands and waterbodies, and aquatic habitats along the river itself. Hence, the naturally dynamic landscape along the Sacramento River readily provided the key habitat elements required for bank swallow reproductive success (Figure 7-5).

We hypothesize that historical bank swallow population levels were higher than they are today, but the overall distribution of the species was probably similar, with higher densities throughout its current range from Verona to Redding. The greatest numbers of burrows and colonies would likely have occurred in the most actively meandering reach, from Red Bluff to Colusa (RM 243–143).

The migratory behavior and timing of breeding of bank swallows has presumably adapted in response to a number of factors including food availability and ambient temperature. Although it is difficult to know precisely what gave rise to the timing of bank swallow breeding along the Sacramento River, it matches the natural flood-pulse cycle (Figure 7-6), such that the species avoids nesting in the winter, when high flows were most common and therefore most likely to erode or inundate colony sites. Nest construction began in the spring when, in most years, flows were reduced to suitable levels after leaving an abundance of recently eroded banks of suitable height, soil texture, and proximity to foraging sites.

7.3.1.1 Site fidelity and metapopulation dynamics

The intermediate level of site fidelity exhibited by bank swallows (Ehrlich et al. 1988, Garrison 1999) seems well-suited to dynamic alluvial river-floodplain landscapes. Complete fidelity to specific colony sites would not be a successful strategy for bank swallows, because large floods have the potential to completely eliminate individual sites by erosion, and because vegetation can encroach and ectoparasites at burrow sites can build up after a series of stable years. Slightly relaxed site fidelity, with birds returning to the general vicinity of the previous colony site (e.g., to an actively migrating reach of the river), is probably a more optimal strategy. This would allow birds to either return to previous colony sites when they are still suitable, or alternatively relocate to nearby freshly eroded banks. Under this system, a majority of intraspecific interactions during breeding season would be with members of the current colony, but there would also generally be some interaction with members of other nearby colonies. Increased interaction among local colonies might be particularly prevalent in breeding seasons that followed large winter floods, which would shift the mosaic of suitable banks and thus force birds to search for new nesting sites.

Under such a system individual colonies would act as subpopulations within a larger overall population along the river corridor, with limited movement among colonies. These considerations imply that metapopulation dynamics may play an important role in the ecology and viability of bank swallows along the Sacramento River, both historically and under current conditions (see Moffatt et al. 2005 and Section 7.5 for further discussion). Individual colonies along the Sacramento River typically last for only a few years, with an upper limit of 5–7 years (B. Garrison, pers. comm., 2005; Moffatt et al. 2005). Therefore, persistence of the larger metapopulation along the river corridor depends on a balance between local extinction and colonization, where extinctions are defined as the disappearance of a colony from a site and colonizations occur when birds begin nesting at an unoccupied site (Moffatt et al. 2005). Bank swallows along the Sacramento River appear to conform to a relatively complex metapopulation structure in which all patches (i.e., existing and potential colony sites) are not equal and the probabilities of colonization and extinctions vary through time and space (Moffatt et al. 2005).

Although we lack good historical data on bank swallow populations in the state, available evidence indicates that bank swallows were common throughout the Sacramento Valley and in many other parts of California prior to widespread channelization and flow regulation of lowland alluvial rivers. Natural landscape conditions and processes created a dynamic landscape, to which the bank swallow is apparently well adapted. Local colony extinctions and new colonizations were common, but the larger metapopulation along the Sacramento River was likely robust and probably served as a source of colonizers for sites along the Sacramento River's major tributaries and possibly elsewhere in the Central Valley and beyond. Such a well-distributed and robust metapopulation structure in the Sacramento Valley has probably been quite resilient to local disturbances from flood or drought. It can also be reasonably assumed that, under historical conditions (pre-1850), habitat quality and quantity along migratory pathways and in wintering areas in Central and South America were generally high and capable of supporting bank swallow populations at higher levels than are currently observed.

7.4 Conceptual Model of Bank Swallow Habitat Dynamics

To shed light on how human-induced changes have affected bank swallow breeding habitats, it is necessary to first consider how key bank swallow habitats might have evolved under natural conditions. Figure 7-7 summarizes a conceptual model of breeding habitat dynamics under natural conditions, highlighting the importance of progressive meander migration during periods in which nests are not occupied by bank swallows. Much of the supporting background material on geomorphic processes that was considered in development of this conceptual model is presented in Chapter 3.

The need for periodic renewal of nesting habitat is dictated by the progressive decline in burrow quality, due to erosion that reduces bank slopes (and thus provides easier access for predators) and infestation by fleas and other nest ectoparasites. Renewal needs are met when the main channel meander bend is eroded by flows that overcome the bank's resistance to erosion (Figure 7-7)—which is set primarily by the strength of the bank materials and modified, in some cases, by root strength of vegetation.

Available data on migration rates and the distribution of bank swallow populations confirm that areas of rapid migration generally support more bank swallows (Figure 7-8 and Table 7-5). In a comparison of stable and active reaches (with a threshold migration rate of 6.6 ft/yr (2 m/yr) separating the two categories), active reaches clearly support more colonies and burrows per unit of centerline length (Table 7-5). Moreover, the densities of colonies and burrows both appear to increase systematically with increasing meander migration rates (Figure 7-8).

Table 7-5. Migration rates and densities of bank swallow burrows and colonies*.

Reach #	Type	Upper RM	Lower RM	1997 centerline length		Average migration rate 1946–1997		Average burrow density		Average colony density	
				mi	km	ft/yr	m/yr	burrow/mi	burrows/km	colonies/mi	colonies/km
1	A	243	232	10.0	16.1	11.0±2.5	3.4±0.8	264	164	0.53	0.33
2	S	232	229	2.8	4.5	4.3±1.2	1.3±0.4	82	51	0.47	0.29
3	A	229	218.5	10.4	16.8	12.6±2.1	3.9±0.7	208	129	0.40	0.25
4	S	218.5	216.5	2.1	3.3	1.5	0.5	0	0	0.00	0.00
5	A	216.5	201	13.0	20.9	12.6±1.8	3.9±0.6	161	100	0.37	0.23
6	S	201	198.5	2.5	4		2.0	40	25	0.18	0.11
7	A	198.5	178	22.3	35.8	16.1±1.6	4.9±0.5	183	114	0.39	0.24
8	S	178	176	2.0	3.2	1.3	0.4	64	40	0.29	0.18
9	A	176	169	6.7	10.8	15.9±3.0	4.9±0.9	275	171	0.72	0.45
10	S	169	167.5	1.4	2.2	NA	NA	39	24	0.42	0.26
11	A	167.5	165	2.3	3.7	17.4±6.2	5.3±1.9	320	203	0.81	0.50
12	S	165	164	1.1	1.8	3.0	0.9	15	9	0.13	0.08
13	A	164	155	9.1	14.6	14.6±3.5	4.5±1.1	195	121	0.56	0.35
14	S	155	151	4.7	7.5	NA	NA	23	14	0.18	0.11
15	A	151	144	5.4	8.7	13.5±2.3	4.1±0.7	122	76	0.43	0.27

Uncertainties, reported where available, are standard errors of the mean.
*Burrow and colony densities averaged by reach for 15 active (A) and stable (S) reaches (as defined by Constantine et al., unpublished) using data from the CDFG annual bank swallow surveys of 1998–2004 (Hight 2000, Schlorff 1998, 1999, 2000, 2001, 2002, 2003, 2004). Meander migration rates are averaged over the post-dam interval (1946–1997) for each of the 15 reaches (from Constantine et al., unpublished).

In general it is expected that the erosivity of flows increases with meander bend sinuosity, and that resistance to erosion increases with soil cohesion, which depends on clay and silt content. Hence, progressive migration tends to be most pronounced in highly sinuous areas that have sandy banks. Over time, progressive migration has the self-reinforcing effect of increasing sinuosity, which also increases the probability that channel cutoffs will occur and thus create new main channels (Figure 7-7). Chute cutoff can quickly create long new stretches of suitable bank swallow habitat (if newly exposed bank soils have compositions suitable for burrow construction), but may strand existing habitat in the old main channel, making further renewal there unlikely, and leading to eventual abandonment of established bank swallow colonies. Chute cutoffs generally reduce channel sinuosity, which in turn reduces meander migration rates locally (Chapter 3). Hence, rates of renewal of bank swallow breeding habitats are expected to generally be reduced over the long-term in areas that are affected by chute cutoffs.

Progressive migration can alternatively be beneficial or detrimental, with the timing of the high flows that cause erosion being the crucial determining factor. As discussed above, high flows during winter (when bank swallows are absent from the river) renew nesting habitat through bank erosion and are typically beneficial. Moffatt et al. (2005) found a positive correlation between winter peak flows and bank swallow metapopulation size. Conversely, high flows during late spring or early summer are thought to be most detrimental (Figure 7-7), causing erosion of the steep outside bends of meanders, which breeding bank swallows prefer (Figure 7-8). This can result in full or partial loss of nest burrows located in the eroding banks, and thus cause high mortality of bank swallow eggs, incubators, and nestlings. High flows during breeding season can also cause inundation of nests in colonies where burrows are close to river level (Figure 7-7). The widely observed preferred burrow height of 6.6 ft (2 m) or more (above the base-flow water surface elevation) (Humphrey and Garrison 1987, Garrison 1999) generally precludes significant effects of inundation during summer flows, which exhibit relatively small fluctuations in stage. Under historical conditions, inundation would have been most detrimental during late spring storms.

Rates of change of flow are important additional considerations for determining potential success of bank swallow breeding. Many banks have been observed to fail during the receding stages of flood peaks, when saturated bank soils that are stranded above the water level succumb to high internal pore pressures and fail catastrophically, undermining overlying material and causing bank collapse (Buer 1994). In areas that are heavily populated by bank swallows, mortality rates are likely to be very high due to bank collapses such as these. However, the frequency of bank collapse during bank swallow breeding season under current flow management practices is likely low enough (i.e., very rare and localized) that effects on the overall bank swallow population along the Sacramento River are relatively minor. Although it is unlikely, if a rare large magnitude flood event did occur during the breeding season it could be catastrophic, as it might result in the substantial loss of that year's cohort.

7.5 Effects of Changes in Bank Swallow Habitat

7.5.1 Changes in the frequency and magnitude of winter flows

As discussed in Chapter 2, there have been several notable changes in the frequency and magnitude of winter flows due to regulation of the Sacramento River. For example, the magnitude of peak winter discharges has been reduced by roughly 50% (Kondolf et al. 2000), and the flood with recurrence interval equal to 1.5 years—often roughly equal to a river's bankfull or "formative" discharge under natural conditions—has been reduced from an estimated 86,000 cfs to roughly 61,000 cfs in the post-dam era (Kondolf et al. 2000). The implication of these and other human-induced changes in the frequency and magnitude of flow is that there has presumably been a reduction in the frequency of flows capable of causing widespread erosion of vertical banks. According to one estimate, for example, flows required to induce measurable lateral change in bank position (i.e., progressive meander migration) may occur only once every five to ten years rather than once every other year or so, as they did in the pre-dam era (Micheli and Larsen, in preparation).

Hence, the river's ability to create new bank swallow habitat should have been affected by a reduced frequency and integrated magnitude of bank erosion in the post-dam era. Yet the expected decreases in rates of bank erosion and channel migration have not been observed (Micheli et al. 2004; Constantine et al., unpublished). If anything, average migration rates appear to have increased slightly in the post-dam era in most cases for individual reaches (Figure 7-7, and discussion in Chapter 3; Constantine et al., unpublished). Moreover, when static (i.e., non-migrating) stretches of river are excluded from the analysis, the overall average migration rate for the river as whole (from RM 243–143) has nearly doubled in the post-dam interval (Micheli et al. 2004). This indicates that migration rates in stretches of river that have actually shifted laterally are now much faster than they were in the pre-dam interval.

There is some indication (see Chapter 3) that the relatively high migration rates of the post-dam era may have been maintained—despite changes in flow that would be expected to decrease migration rates—by effects of progressive conversion of the floodplain from riparian forest to agricultural lands. Removal of riparian forests in highly sinuous bends is thought to promote meander bend cutoff (K. Buer, pers. comm., 2005), by increasing susceptibility to cutoff and progressive lateral migration over the short term. The observed increase in migration rates in the post-dam era, although small, may be due to an increase in cutoff frequency that has resulted from an increase in removal of riparian forests from the floodplain.

If the inferred increase in meander migration rates is due to increases in the relative frequency (or importance) of meander bend cutoff (compared to progressive migration), then there may have been a net detrimental effect on bank swallow breeding habitat. Cutoffs rapidly create long new stretches of fresh banks, which may good for bank swallows initially if other nesting habitats requirements are met along the new main channel. But as discussed in Section 7.3, cutoffs may also reduce habitat renewal rates in the old main channel and moreover cause reductions in channel sinuousity that may suppress progressive migration rates in the new main channel. Hence, the net effect of increased meander migration rates due to increased cutoff frequency may be negative over the long-term for the bank swallow population along the middle Sacramento River, although there is much uncertainty and further study of this issue is warranted.

There is some indication, based on analysis of metapopulation dynamics (Moffat et al. 2005), that the probability of both colonization and extinction of bank swallow colonies is positively correlated with maximum river discharge in the preceding year (Figure 7-9). This highlights the tight coupling of bank swallow habitat and the geomorphic processes of bank erosion, which are regulated for the most part by the frequency and magnitude of peak discharges.

7.5.2 Changes in the magnitude and rate of change of summer flow

As discussed in greater detail in Chapter 2, flow regulation on the Sacramento River has increased summer baseflows by roughly 100% to satisfy downstream irrigation demands (Kondolf et al. 2000, Snowden 2002). Flow regulation has also increased the number of summer flow reversals. Potential implications of changes in the pattern of summer flows for bank swallows include the possibilities of increased risk of direct mortality and disruption of pair bonding due to inundation and bank failure, if high flows and rapid flow reversals occur when birds are breeding (i.e., from March–July). It is unclear just how important these potential risks to bank swallows are under the current flow management regime (see below), however such risks should be considered if changes in flow management are proposed in the future that might increase the magnitude or frequency of high flows and rapid flow reversals during the breeding season.

7.5.2.1 Flow magnitude

The increase in summer baseflows has probably never been big enough, by itself, to cause inundation, because most bank swallow nests are typically safely located 6.6 ft (2 m) or more above the summer baseflow water level. On the other hand the possibility that high summer baseflows can lead to bank collapse and colony failure cannot be ruled out. Bank erosion thresholds vary widely depending on local conditions. In some cases the threshold may be well below 12,000 cfs, which is often exceeded during the augmented summer flow period, when bank swallows are nesting. For example, as noted in Chapter 3 (Table 3-6) analyses of field-based data suggest that bank erosion on the Sacramento River can be initiated at flows as low as 7,500 cfs near Princeton—with significant (i.e., order factor of four) variability from site-to-site depending on local conditions (Kondolf et al. 2000). Direct field observations have confirmed that bank erosion may begin at flows as low as 10,600 cfs at some sites (Larsen et al., unpublished). In their bank erosion studies, CDWR (Buer 1994, 1995; Klinesteker 1998) suggested that the threshold of bank erosion at most sites was above 13,000 cfs. Buer (1994) went on to suggest that bank erosion rates might increase exponentially with discharge up to bankfull discharge (estimated to currently be about 88,000 cfs; Thomas 2000). Summer base flows were not observed to be strongly correlated with field observations of bank erosion events in the CDWR studies. Moreover, field observations suggest that only localized incidents of bank erosion occur at flows less than 60,000 cfs, whereas more widespread erosion may occur at higher flows (K. Buer, pers. comm., 2000).

There have been at least two documented accounts of colony failure due to flow-related bank collapses. The first occurred at RM 195.0 in May 1988, when a colony of 907 active burrows was reduced to 283 due to bank collapse following flows that peaked at just 14,300 cfs (Figure 7-10; B. Garrison, pers. comm., 2005).

The second account of flow-related bank collapse occurred at RM 166.3 (near Princeton); a colony of 772 burrows was washed away during a late spring 1993 storm, in the first week of June (B. Garrison, pers. comm., 2005).

These two observations highlight a potentially significant indirect mechanism of bank failure: If late spring or early summer storms bring high flow inputs from unregulated tributaries, and baseflow on the mainstem is already unnaturally high (due to regulation), then increases in flow due to storm inputs might be high enough to cause failure or inundation that would not have happened in the absence of regulation. This appears to have occurred on the mainstem Sacramento River in at least one instance due to a high summer flow from tributaries.

The corresponding instantaneous flow hydrograph for the nearby Butte City gauge is shown in Figure 7-11. Discharge nearly tripled during the interval of bank collapse, with an instantaneous peak of 33,100 cfs. Sandy, easily eroded banks are characteristic of the reach in the vicinity of the RM 166.3 failure site (K. Buer, pers. comm., 2005). In one study (Buer 1994) the river's highest bank erosion rates were reported to occur nearby. Hence the site may not be representative of the river at large. Even so, the hydrographs shown in Figure 7-11 highlight the potential importance of flow regulation on summer bank collapses; the peak flow release from Keswick Dam occurred well after the first flow peak arrived at Butte City and moreover apparently contributed to a delayed second peak at the bank swallow collapse site—which therefore experienced an extended period of high flows (possibly contributing to the observed bank collapse). However, it is likely that the initial peak discharge due to tributary inputs was the primary cause of the bank failure and loss of burrows.

Analysis of the hydrographs in Figure 7-11 highlights a potential means for managing the risk of flow-related collapses of bank swallow colonies. If the peak flow release from Keswick during the 1993 storm had been delayed by a few more days (assuming reservoir capacity would have allowed it), it might have been possible to reduce the length of time that flows in the vicinity of Butte City (RM 169) were above the bank erosion threshold, and thus reduce the chance that the colony at RM 166.3 was affected by the flow event. It is unclear how often erosion events such as the one observed in 1993 occur. Nevertheless, it seems clear that careful management of flow releases may help reduce bank swallow mortality during spring and summer storms.

7.5.2.2 Flow reversals

Under regulated conditions, the number of flow reversals in the summer has increased. One potential implication for bank swallows is an increased risk of minor bank failures caused by reduced bank stability when retreating stage strands partially (or completely) saturated soils above the water line (Buer 1994). The effects of rapid flow reversal are probably exacerbated by the effects of high flow that precede them (with undermining of banks due to mobilization of material at bank toes). In general, rates of bank migration presumably reach a maximum on the falling limb of the hydrograph, following periods of bankfull (or higher) flow, when shear stresses are high at the bank toe and falling stage creates positive pore pressures that promote bank failure.

Minor bank failures caused by flow reversals (or the combined effect of flow reversals and high flows) can cause direct mortality, if they occur at colony sites when eggs or chicks are in the burrows. Minor bank failures can also convert safe, near-vertical banks into unsafe banks by generating a ramp of failed material that reduces the amount of bank that is high enough to afford protection from predators. Both "vertical" and "ramped" banks occur along the Sacramento River, but their relative abundance and importance for bank swallow nesting is not known although bank swallows do nest in both types of banks (B. Garrison and R. Schlorff, pers. comm., 2005). Also unknown is whether “vertical” banks significantly differ from “ramped” banks in susceptibility to erosion by high flows during breeding season.

The effect of increased summer flow reversals on bank swallow populations is unknown, although we hypothesize that is less important than the effects of peak flows. More frequent monitoring of bank swallow populations along the Sacramento River throughout the breeding season might help shed additional light on the effects of flow reversals on bank failure at colony sites. If it is determined that flow reversals do cause a significant impact to nesting bank swallows, it is possible that the adverse effects could be mitigated for by reducing rates of reversal, and thus allowing pore pressures within banks to decline slowly, without causing bank instability.

7.5.2.3 Changes in rates and styles of lateral migration

Assuming that the overall discharge of water down the Sacramento River is roughly constant over the long term, the regulation-related increases in summer baseflow presumably reduce the percentage of time that flows overtop the river's banks. That this is true can be verified by consideration of a simple mass balance of discharge for the river:

- the total volume of water is fixed
- flows during summer are increased, but not enough that they can over top the banks
- The amount of water available for overbank flows in winter must therefore be lower.

This is confirmed by indications that bankfull discharge is much less common now than it was in the pre-dam era. This has also presumably reduced the overall probability of meander bend cutoff, and conversely may have increased progressive migration rates. The rationale for this hypothesis is that the increase in summer base flows has increased shear stresses enough to increase cumulative effective stream power, such that rates of progressive migration have been accelerated over the long term. This would have indirect implications for cutoff migration processes, because progressive migration regulates channel planform curvature, and thus influences the probability of cutoff initiation at any given point on the river.

7.5.3 Effects of bank armoring activities

7.5.3.1 Reduction in suitable nesting sites caused by bank protection

The installation of riprap and concrete in bank armoring activities can have the immediate effect of reducing the availability of sufficiently steep, suitably textured habitat for bank swallow nesting colonies. While bank revetment structures are rarely 100% effective at halting erosion, and often only relocate the problem, they are generally effective at reducing meander migration rates locally. If soils in the affected reach were suitable for nest construction, then the bank protection would lead to an overall decrease in the rate of breeding habitat renewal.

Land owners had begun implementing localized riprap style bank armoring projects by as early as the mid to late 1800s, but the vast majority of bank protection structures were installed by the Army Corps from the mid 1960s through the 1980s, when an estimated 34% of the channel margin between RM 194–143.5 was covered with riprap or concrete rubble (Greco et al, unpublished [2006a]), based on USACE 1986). Overall, an estimated 48% of the channel from Red Bluff to Colusa (RM 243–143) is now covered by riprap on at least one side (Larsen and Greco 2002, S. Greco, unpublished data). Figure 7-12 provides an example of the relative amount and distribution of riprap and other bank armoring currently found in the reach between RM 229–218.

Bank protection has been preferentially applied to actively migrating bends which would otherwise be among the most suitable sites for bank swallow nests. Hence, it is likely that bank protection has eliminated substantially more than 48% of potential nesting sites between Red Bluff and Colusa. Plans for new bank protection projects on the Sacramento River continue to be developed. If implemented, these projects would further reduce available habitat, with an extremely detrimental cumulative effect on the bank swallow population (Schlorff 2004). Given the amount of habitat lost to date due to bank protection and current population numbers, the cumulative effect of any net increase in bank protection along the middle Sacramento River that further reduces nesting habitat is likely to jeopardize the viability of the Sacramento River bank swallow population (Schlorff 1997).

7.5.3.2 Direct effects of construction

If construction activities occur during breeding season, bank protection can cause direct mortality of bank swallows—particularly to eggs and nestlings. The construction timing of state and federally sponsored bank protection projects is now regulated by the stipulations of the Migratory Bird Treaty Act and the California Endangered Species Act (Schlorff 2004), but before this was the case (i.e., prior to 1986), direct mortality and loss of entire colonies was observed to occur during bank protection installation in a number of cases. In 1985 alone, for example, bank protection construction activities conducted at the height of the breeding season destroyed the habitat at 3 breeding colonies along the river, which included approximately 1300 burrows (representing and estimated minimum of 725 breeding pairs) (CDFG 1992, Garrison and McKernan 1994). Another 6 colony sites, which contained approximately 2000 burrows, were eliminated by bank protection activities during 1986 and 1987 (CDFG 1992, Garrison et al. 1989). Direct mortality due to bank protection activities is now greatly reduced compared to pre-1986 levels (due to regulatory enforcement), but some unofficial bank protection projects still continue and have the potential to locally affect bank swallows (Schlorff 2004).

7.5.4 Human-induced changes in surrounding landscapes

7.5.4.1 Conversion of land cover for agriculture and other human uses

Losses and reductions in the area of grasslands, lakes, ponds, rivers and streams, meadows, fields, pastures, bogs, forests and woodlands surrounding colonies have affected foraging area for bank swallows. Especially significant declines in foraging habitat are thought to have occurred as a result of the conversion of native grasslands to developed land. Besides being a productive habitat for insect prey, grasslands tend to produce abundant thermal updrafts, which help lift airborne prey and thus expose them to easier capture by bank swallows and other foraging birds (Drake and Farrow 1988). The total loss of grassland habitat is difficult to quantify in the absence of historical data and in any case would be difficult to interpret as a net effect on bank swallows, without information about proximity of historical grasslands to colony sites.

Forests and woodlands may have been among the least productive foraging habitats for bank swallows, due to relatively low abundance of aerial insects, obstructed flight paths limiting aerial foraging by swallows, and relatively stagnant overlying columns of air. Herbaceous patches within forests would have been more productive, but their extent was probably too limited to substantially reduce extinction risks of nearby bank swallow colonies. The clearing of riparian forests by humans may have nevertheless had substantial indirect effects on bank swallow populations locally, particularly in instances in which forest clearing promoted the initiation of channel cutoff processes in bends that had previously supported rapid progressive migration rates and abundant bank swallows (Figure 7-8). As noted in Section 7.3 (and Figure 7-7), cutoffs quickly create long new stretches of bank, which may be good at first for bank swallows if soils are suitable. But cutoffs also strand any existing colonies in the old main channel, making further habitat-restoring erosion there unlikely. Moreover, because a new cutoff has reduced sinuosity (by definition) relative to the old main channel, it is likely to have a relatively low migration rate, making any new habitat created in the cutoff process ephemeral at best. Anecdotal accounts (K. Buer, pers. comm., 2005) and analyses of historical aerial photos (Micheli et al. 2004) confirm that channel cutoffs on the Sacramento River have often been immediately preceded by the clearing of riparian forest vegetation in the cutoff bend. Moreover, as discussed in Chapter 3, channel sinuosity in cutoff bends appears to have declined slightly since the 1940s when the dams were constructed and the conversion of the natural floodplains to agricultural uses intensified along the Sacramento River (Table 3-8). The effects of human-induced cutoffs and reductions in sinuosity on local bank swallow populations are unknown due to a lack of historical data on bank swallow distributions. However, it can be reasonably assumed that if sinuosity declines locally as a result of anthropogenic disturbance, then the overall average rate of migration is likely to decline locally as well due to both a reduced propensity for meander bend cutoff and a reduced rate of progressive migration. This would affect the rate of renewal of existing bank swallow habitat and lead to a less dynamic channel. Continued monitoring of bank swallow populations in the vicinity of incipient cutoffs should help shed light on the importance of these effects.

7.5.4.2 Pesticide use

Pesticide use does not appear to have any substantial direct effects on bank swallows. For example, studies of eggshell thickness have not detected any of the systematic thinning that would be symptomatic of potentially important pesticide-related effects (Schlorff 1997). On the other hand, heavy pesticide use can have the indirect effect of reducing prey abundance over agricultural lands and adjacent grasslands (due to spillover effects), with possible but difficult to quantify implications for extinction rates of nearby bank swallow colonies (Moffatt et al. 2005).

7.5.4.3 Increases in predator abundance

Human land use activities in the river corridor have led to the introduction of non-native predators, such as Norway rats and black rats, and appear to have facilitated an increase in the local abundance of native predators such as raccoons. It is possible that increases in the abundance of these predators in the river corridor has led to some increase in bank swallow mortality or nest failure, but no studies have yet been conducted to test this hypothesis.

7.5.5 Metapopulation dynamics and population viability

Bank swallow numbers vary both spatially and temporally (Figures 7-1, 7-2). Longitudinal variability along the river corridor occurs at multiple spatial scales, including reaches and subreaches ranging in length from 10 to 100 river miles (Figure 7-2), and at more local scales such as the 5-mile sections shown in Figures 7-13 and 7-14. Although availability of nesting habitat is generally considered to be a major factor affecting the size and distribution of bank swallow breeding populations throughout the Northern Hemisphere (Cramp et al. 1988 and Turner and Rose 1989, both as cited in Garrison 1999), other factors (e.g., predation, food supply, parasitism, competition, severe weather) may also affect local population dynamics. It is not always clear what is driving spatial variability in any given year, but heterogeneity in bank conditions and food supply (i.e., high quality foraging sites) are potential contributing factors that vary naturally and that can be affected by anthropogenic land and water management actions. Local variability in air temperature and precipitation may also be important, particularly as local weather can affect availability of aerial insect prey (Taylor 1963, Speakman et al 2000). Interannual variability in wintering and migratory conditions may also play an important role.

For example, drought conditions in North African wintering habitats may have contributed to declines in several European breeding populations (see Jones 1987, and summary in Garrison 1999), and severe weather has been known to cause significant mortality during migration (Sealy 1966, as cited in Garrison 1999).

Local variation in the number of colonies and burrows per 5-mile section can be observed by comparing values for the early survey years (1986–1988) when the population was at a peak, to the partial recovery period (1998–2000) and the more recent apparent stabilization period (2002–2004). Even within each 3-year period there is a lot of year-to-year variability evident, although the variation among the three 3-year periods is more pronounced. Some sections with moderate numbers in at least 2 out of 3 years during the early period (e.g., RM 150 in Figure 7-13 and 7-14) show a notable reduction 10 years later and local extinction by 2002. An adjacent section (RM 145) showed a similar general reduction between the first two periods, but had bounced back by the more recent period. It is unclear how well changes in local bank habitat and variation in flows and weather correlate with this observed degree of spatial and temporal variability, but a variety of causal linkages can be hypothesized. For example, a local rebound in bank swallow populations could occur if birds from an adjacent colony began to utilize the newly evacuated reach when they returned to the area at the beginning of the next breeding season. There appears to be little within-breeding season movement among colonies, so observed between year variability is most likely a response of returning adults tracking changes in habitat location, abundance and quality at the beginning of each breeding season (B. Garrison, pers. comm., 2006).

The recent partial recovery in population trends is not fully understood (see Figure 7-1, 7-2 and Section 7.1). One hypothesis is that the decline from 1986 levels and subsequent partial recovery since 1998 may have been related to variations in rainfall and bank erosion patterns, which can affect habitat quality. It has been noted, for example, that the steep decline from the late 1980s through the mid-1990s is roughly coincident with a period of extended drought (Schlorff 2004) in which there was a sharp reduction in the frequency of flows that exceed the threshold for widespread bank erosion, equal to roughly 60,000 cfs according to anecdotal accounts (K. Buer, GOOD Geotechnical Consultants, pers. comm., 2005) (Figure 7-15).

Another hypothesis is that shifts in population trends are not due to changes in local habitat conditions but instead have been caused by changes in wintering grounds in north-central South America. As discussed above, drought in the winter habitats in North Africa appear to have caused declines in a number of European breeding populations.

The recent stabilization in Sacramento Valley bank swallow numbers is encouraging, but the average number of colonies in recent years (2000–2004) is still lower than it was in earlier surveys (Figure 7-1), implying that there may have been a net loss in suitable habitat. Large (> 1000 nests) colonies are an indicator of overall population health and are thought to provide sources of colonists for re-population of areas that may become abandoned from time to time due to natural variations in habitat availability. In recent years there have been a small number of large colonies present each year (Figure 7-16). For example, in 2000 and 2004 there were 5 and 4 large colonies present, respectively, while in 1998, the beginning of the partial recovery, there was only 1 large colony present. There was a general shift in the distribution of colony sizes toward larger colonies between 1998 and 2000, however, by 2004 the distribution had shifted back to an intermediate distribution (Figure 7-16).

7.6 Conceptual Model of Current Conditions

Bank swallow abundance in the Sacramento Valley has declined substantially relative to historical conditions. The main causes of the decline appear to be loss and alteration of suitable breeding habitat. The current distribution of bank swallows along the middle Sacramento River is probably similar to the historical distribution, albeit at reduced densities, especially downstream of Colusa. The 100-mi (161-km) reach from Red Bluff to Colusa (RM 243–143) is currently the center of the Sacramento Valley population, and the critical production zone for bank swallows in California. As such, the long-term viability of this threatened species in the state is dependent upon maintaining and enhancing the Sacramento River population.

The largest contributor to the decline has probably been direct loss of nesting habitat due to bank armoring, coupled with direct mortality of whole colonies during bank armoring construction activities that occurred during the nesting season. Proposed bank armoring projects for the river continue to threaten existing and potential future bank swallow breeding habitat. Because bank armor (e.g., riprap) is usually placed on the most actively eroding banks, it results in an immediate and disproportionate direct loss of suitable nesting habitat. In addition, bank armoring alters spatial and temporal patterns of bank erosion, resulting in potentially complicated long-term indirect effects on the renewal of the supply of suitable banks in the immediate vicinity of the armored bank.

Human development of native riparian habitats—particularly grasslands and wetlands—for agricultural or other uses has most probably been a significant contributing factor in the bank swallow decline. Native grasslands and wetlands provide highly productive foraging habitat, with abundant insect prey and thermal uplifts suitable for efficient aerial foraging by bank swallows. In contrast, agricultural fields (e.g., with orchards and row crops) provide altered physical habitat and are typically intensively managed with pesticides, such that potential prey are scarce. The net effect of land development on bank swallows has been a decrease in the number and size of high-quality grassland and wetland foraging habitats in close proximity (i.e., within roughly 660 ft [200 m]) to suitable nesting sites. However, we hypothesize that the net effect of riparian foraging habitat alteration ahs been much less important than the direct loss of nesting habitat due to bank armoring.

Combined effects of Shasta Dam and various flood and erosion control practices would be expected to alter the long-term rates of lateral river migration, and thus affect the rate of renewal of bank swallow habitat. However, the net effect of human-induced changes in flow and sediment transport on bank swallow abundance and population dynamics is not well understood. In general, any long-term trend of reduced progressive meander migration rates would be expected to have significant negative effects on the bank swallow population. Available data suggest that patterns of flow, sediment transport and erodibility have shifted such that there is a lower threshold for channel cutoff at many of the Sacramento River's meander bends, producing cutoff bends with lower sinuosity and a lower contribution of progressive migration to the river's lateral migration, compared to historical conditions. The increased cutoff rate and decreased sinuosity of cutoff bends may cause potentially significant adverse impacts on long-term viability of the bank swallow population; short-term benefits of new banks created by increased cutoff would probably be more than offset by detrimental effects of reductions in the frequency and magnitude of progressive bank erosion, which is needed to periodically “renew” nesting sites and thus maintain freshly eroded conditions required for high habitat suitability.

Alterations in summer flow conditions may also be affecting bank swallows to some degree. Increased summer base flows result in increased average river stage during the nesting season, which might make a few of the lower bank sites unsuitable due to reduced relative bank height (height above the water surface). However, a recent analysis of the heights of vertical banks along the river indicates that average bank heights along most sections of the river from Woodson Bridge to Colusa (RM 218–143) are 6.6 ft (2 m) or greater (Greco et al., 2006b); Figure 7-17), indicating that they would therefore fall into the highly suitable (SI=1.0) bank height category (Figure 7-3c). This suggests that bank height may not be a key limiting factor—unless many of the banks of suitable height have been riprapped (the analysis of Greco et al. (2006b) did not differentiate between unprotected from protected banks).

Bank swallow nesting habitat is extremely ephemeral due to the interaction between the friable soils needed for burrow excavation and the erosive forces needed to maintain vertical faces at the colony sites. Without some erosion, human-caused or otherwise, the vertical faces quickly collapse and break down, thereby becoming unsuitable for nesting. Colonies do not typically occur in every bank or bluff that is suitable, nor do burrows occupy all suitable locations within an individual colony site. There is also considerable turnover in colony sites from year to year. On the Sacramento River, bank swallows generally nest in 25–33% of the total number of banks that are suitable for nesting (according to criteria outlined in HSI model) in any given year; populations apparently require some habitat surplus in order to remain viable over the long-term (Garrison 1998; B. Garrison and R. Schlorff, pers. comm., 2005), although we do not know how much surplus habitat is needed each year to promote recovery and maintenance of a viable population. This conclusion is supported by a recent theoretical metapopulation analysis of colonial breeders which indicated that site fidelity combined with local and especially global density-dependent factors could lead to low habitat patch (potential colony site) occupancy rates even under equilibrium conditions (Matthiopoulos et al. 2005).

The cumulative reduction in nesting habitat quantity and quality compared to historical conditions has led to a decrease in the number of colonies and total abundance of the Sacramento Valley population of bank swallows, and likely reduced its resiliency and resistance to disturbance. The recent level of 8,000–10,000 breeding pairs likely has a substantial risk of falling to relatively low numbers within 50 years (based on results of the population viability analysis reported in Buechner 1992 and CDFG 1992). The current numbers are not large enough to ensure long-term persistence of a large, robust population (CDFG 1992, Moffatt et al. 2005). Without substantial action to increase suitable nesting habitat and enhance population levels, the Sacramento Valley population remains at risk. The risk may be even greater if degradation in winter habitat in South and Central America has occurred or occurs in the future, although we currently have no data on this issue.

7.7 Management Implications and Key Hypotheses and Uncertainties

This section synthesizes the available information on bank swallow ecology and habitat forming processes to identify potential management actions to preserve and enhance bank swallow populations and habitat along the middle Sacramento River. Because the Sacramento River population is so essential to statewide recovery planning for this species, we first review some of the management strategies and recommendations presented in the state bank swallow recovery plan (CDFG 1992).

Management actions directed at maintaining and enhancing the bank swallow population along the middle Sacramento River should also directly benefit a number of other native terrestrial wildlife species that are associated with eroding banks and bluffs along California lowland alluvial river systems, including the northern rough-winged swallow, black phoebe (*Sayornis* *nigricans*), and belted kingfisher (Garrison 1998). The natural processes of channel meander migration and bank erosion also benefit a variety of native aquatic species, and promote the establishment of new stands of riparian vegetation (see Chapter 9).

7.7.1 Management guidance provided by the Bank Swallow Recovery Plan

Management strategies outlined in the 1992 Bank Swallow Recovery Plan (CFDG 1992) included the protection, enhancement, and maintenance of natural habitats, and an evaluation of the feasibility of artificial habitat construction. Management alternatives emphasized reducing impacts to natural bank habitats, developing a set-back levee/meander belt system, and addressing habitat needs of the bank swallow in existing habitat preserves on the Sacramento River. The ultimate goal of the Bank Swallow Recovery Plan (CFDG 1992) is the maintenance of a self-sustaining wild population. These objectives can be achieved by (1) ensuring that remaining populations do not decline further in either range or abundance, and (2) providing for the preservation of sufficient natural habitat to maintain a viable wild population in perpetuity (CFDG 1992). Any management actions undertaken for bank swallow in California should complement the goals of this Recovery Plan.

Management efforts have rarely been specifically focused on benefiting bank swallows. However, several goals have been proposed to achieve population sustainability in the bank swallow recovery plan, including research needs and management actions (CDFG 1992). Baseline information on population levels combined with habitat inventories are needed to establish habitat objectives. A feasible approach would be to identify all areas of currently and potentially suitable nesting sites in a management area, such as the middle Sacramento River (Garrison 1998).

In the past, artificial and enhanced natural banks were built along the Sacramento River as mitigation for loss of colony sites from flood control projects (Garrison 1991). Bank swallows occupied some of the sites for one to two years following construction, with nestlings produced at levels equivalent to natural sites. In the absence of maintenance, the occupied sites were abandoned within three years, after they had become unsuitable when banks sloughed off, became overgrown with vegetation, or became too hard for burrow construction (Garrison 1991). Artificial banks or habitat enhancements may be successful as a short-term stopgap measure, but the high per-unit-area cost of construction and maintenance dictates that artificial habitat enhancement is not likely to be cost-effective or successful in the long term (Garrison 1998, Garrison and McKernay 1994). Evidence of new, unforeseen problems appearing at artificial bank sites casts further doubt on their likelihood of success. For example, nestlings in artificial banks along the Sacramento River suffered unexpectedly high rates of predation from herons and egrets (Garrison and McKernay 1994). Moreover, without continuous maintenance, ectoparasite loads would probably have developed, causing detrimental effects on nestlings (Garrison and McKernay 1994).

The CDFG recovery plan concludes that, in the absence of extensive conservation of suitable nesting sites over large areas (i.e., combining a strategy of meander migration zone management with one that protects existing high quality habitats), the success of bank swallow preservation will be limited due to their unique population dynamics and the need for natural renewal of nesting sites. Integrating bank swallow habitat protection with broader riparian ecosystem conservation efforts, as is occurring along the Sacramento River as part of the Sacramento River Conservation Area planning process, appears the most promising.

7.7.2 Actions to consider

7.7.2.1 Maintain natural channel migration and bank erosion processes

Assuming the needs of water managers to meet flood control and water delivery requirements can be met, careful consideration should be given to evaluating management and implementation of flows necessary to result in ecologically beneficial bank erosion. Real time assessment and management of tributary inflows, in conjunction with dam releases, is a potential tool that could be used to meet any specific ecological flow targets that might be set for downstream reaches. The discussions above have demonstrated the reduced frequency of flows of a magnitude sufficient to result in bank erosion, however, these flows do still occur. The intent of this management action would be to augment these events if monitoring revealed that this would benefit the ecosystem without unduly jeopardizing key infrastructure, flood control, and water delivery requirements. The appropriate magnitude and frequency of the proposed managed flows will need to be determined, presumably with analysis of how cumulative effective discharge affects migration rates as a first step. The latest (i.e., 2005) CDWR bank erosion survey data, which can be analyzed as soon as the GIS coverage is finalized, should be especially useful for assessment of effects of cumulative effective stream power on migration rates. The release of managed flows to promote bank erosion between November and March, when bank swallows are not present at burrow sites, should be considered. Also required would be a set of target migration rates, with desired conditions determined in careful balance with considerations of potentially adverse effects of increased migration rates and the needs of other Sacramento River focal species. Careful evaluation of effects of such flow releases on the risks of damage to key infrastructures would be required before any such action could be implemented.

7.7.2.2 Modulation of the timing and magnitude of flow from Keswick during spring storms

Managed high flows needed for habitat renewal must occur before the beginning of the breeding season (i.e., before late March) in order to maximize benefits and minimize potential detrimental effects of bank erosion on bank swallows. When possible, flows in the Red Bluff to Colusa reach should be managed during the primary breeding season (April–June) to reduce the risk of substantial colony failure due to bank erosion or nest (burrow) inundation.

It has been suggested that just one day of exceptionally high flow can have significant adverse effects on bank swallows (Moffatt et al. 2005). In reality, the amount of bank erosion and resulting bank swallow mortality (if nests are occupied) or beneficial nest site renewal (if birds are away) are likely to be a cumulative function of discharges that exceed bank erosion thresholds, with individual events having isolated effects and protracted periods of high flow contributing to more extensive, widespread bank erosion. As with the idea of managing winter flows to promote meander migration and bank erosion to create and maintain an abundance of high quality nesting sites (described above in Section 7.7.2.1), any management of flows to benefit bank swallows during the breeding season would require careful consideration of other ecological flow needs (such as cottonwood recruitment flows, see Chapter 9), and flood control and water delivery requirements.

7.7.2.3 Removal of bank armoring to allow bank erosion in reaches with appropriate soils

Removal of bank armoring is likely to increase the availability of suitable nesting habitat. Analysis of metapopulation models suggests that removal of 3–20 % of existing rip-rap could help establish an equilibrium population of bank swallows on the Sacramento River (Moffatt et al. 2005). However, significant recovery for bank swallows via removal of bank armor (riprap or revetment) would probably only be realized if restoration activities were focused along banks that are likely to provide suitable nesting habitat and the population was large enough to expand into the new habitats. As a first approximation, sites could be selected to maximize their HSI (based on the model presented in Section 7.2.4). Another potential constraint on the location of high priority bank revetment removal projects may be proximity to large existing colonies, which might supply an adequately large pool of potential colonizers. The results of levee removal at RM 233 in 1999 (see details below) suggest that removal of riprap and levees may yield rapid benefits in terms of new colony establishment or expansion.

Given the historical habitat losses and reduction in the Sacramento River bank swallow population, no new bank revetment projects should be conducted without thorough assessment of their potential short- and long-term effects on bank swallows and mitigation (avoidance, minimization, and compensation) of adverse impacts. Compensatory mitigation actions should be focused on removal or, possibly, abandonment (i.e., in some cases just stopping maintenance of certain sections of revetment that are no longer needed might allow the river to begin eroding the formerly protected banks) of existing riprap or setting back levees in areas that are (1) not critical for protection of vital human infrastructure, (2) contain suitable soils and channel migration potential for creation of bank swallow nesting habitat, and (3) have landowners willing to cooperate. Mitigation ratios for habitat loss of greater than 1:1 would help contribute towards recovery of the bank swallow population. Creation of a mitigation bank should be explored as a potential strategy that could help maximize short- and long-term benefits to bank swallows by creating larger areas in which natural processes could provide a dynamic landscape with a reliable supply of suitable nesting habitat.

7.7.2.4 Create an expanded meander migration zone by setting back selected levees

As with removal of bank armoring, removing or setting back levees at select sites may help promote progressive meander migration in a way that expands usable habitat for bank swallows. This can lead to immediate benefits for bank swallow populations, as demonstrated on the Sacramento River, after a levee removal and rip-rap retirement project was completed at RM 233 in late fall 1999 (Golet et al. 2003). Erosion induced by winter storms expanded an existing cut bank, and a swallow colony from nearby established itself there in the spring of 2000. The newly established colony, with 2,770 borrows, was the largest on the river that year. It represented a substantial expansion for bank swallows at the site, which had supported just 930 burrows in the previous year. While this single anecdotal account may not be entirely representative of potential gains at other sites, it does suggest that levee setback is a viable option for creating new bank swallow habitat if locations are chosen wisely.

One way to inform selection of levee setback sites and designs is through the use of meander migration modeling (Larsen et al. 2006). In one recent study, the effects of alternative setback scenarios were assessed for a 17 mile-long (28 kilometer-long) reach below Pine Creek (RM 196) (Larsen et al. 2006). Simulations showed that migration rates could be increased by nearly twofold for even the least ambitious (i.e., 330 ft [100 m]) setback scenarios. In segments that have suitable soils and bank heights, this could greatly increase habitat for bank swallows. For more ambitious (i.e., 2,600 ft [800 m]) scenarios, migration was accelerated by eightfold, and was shown to produce conditions that were generally favorable for cutoff---which would produce the off-channel aquatic habitats required by other species, including the western pond turtle (see Chapter 8), and provide potential establishment sites for Fremont cottonwood and other riparian plant species (see Chapter 9). Intermediate setback distances yielded intermediate migration rates and created conditions that were somewhat less favorable for cutoff, relative to the 2,600 ft (800 m) setback option. Taken together, these results suggest that a range of outcomes are possible, and that a variety of species can benefit from management via levee setback. Selecting one option over another will be an issue of balancing the estimated benefits for each individual species of concern with potential costs.

7.7.2.5 Restoration of foraging habitats in the surrounding landscape

While forest restoration has been shown to benefit some neotropical migrant species (Golet et al. 2003, Gardali et al. 2005), rates of colonization and extinction of bank swallows appear to be insensitive to differences in the amount of nearby riparian forest cover (Moffatt et al. 2005). Grassland restoration, on the other hand, would probably be highly effective, reducing extinction rates for nearby colonies (Moffatt et al. 2005). Potential foraging-related benefits might be realized in the absence of grassland restoration through reduction in pesticide use over open areas, such as agricultural fields, although it is not clear whether this would provide significant benefits to swallow populations. It is also possible that other natural vegetation types, particularly savannas and oak woodlands, might provide useful foraging habitat. Further study is needed to evaluate the relative importance of these non-grassland habitats as bank swallow foraging areas (see Section 7.7.4.1). It is clear, however, that restoration of native grassland and riparian scrub and forest habitats would benefit a wide variety of other native species (see Chapter 9, particularly Sections 9.3 and 9.4).

7.7.2.6 Verification of HSI relationships

Some of the elements of the proposed HSI model relationships of Figures 7-4 and 7-5 are somewhat speculative, due to an absence of field data. Verification and refinement of the HSI model by collection of field data should be incorporated into the long-term management strategy for bank swallow conservation. Variables to constrain with new field data are: the maximum suitable distance to nearest grassland (Figure 7-4a), the optimal bank length (Figure 7-3d)—which presumably depends on the currently unknown relationship between suitability and optimal colony size, the range of suitable flows for the breeding season (Figure 7-4c), and the winter migration period (not shown).

The HSI model might be improved overall if the relationship for bank height suitability could be replaced with the potentially more diagnostic relationship between suitability and the heights of burrows above average summer baseflow water surface. Field measurements of individual burrow heights from the Sacramento River would be needed to define such a relationship.

More detailed mapping of floodplain soil types to produce a fine-scale GIS soil texture coverage would allow more accurate modeling of HSI values expected at any specific location subject to future channel migration and bank erosion. This would be an important step towards improving our ability to link physical process models to expected biological responses (see Section 7.7.2.9).

7.7.2.7 Comprehensive surveys of physical parameters and intra- and inter-colony dynamics

An intensive study should be conducted for several years on a representative subset of colonies to measure clutch sizes and nesting success rates (and other demographic data, such as age-specific survival rates and nestling growth rates, related data such as ectoparasite load, and ideally data on diet and foraging locations) and confirm whether data collected in 1986 (Garrison et al. 1987) on burrow occupancy and other factors are consistent with current conditions.

Banding studies at the same subset of colonies to track presence and movement patterns of individuals within and among years should help to improve our understanding of population dynamics, including providing information on immigration and emigration rates and site fidelity in relation to factors such as colony productivity and habitat quality.

7.7.2.8 Modification of long-term bank swallow monitoring program

Continuation of the annual CDFG surveys of colonies along the Sacramento River from Red Bluff to Colusa (RM 243–143) is crucial. However, as a potential modification to the current methodology, researchers should consider increasing the frequency of surveys in the Redding to Red Bluff (RM 292–243) and Colusa to Verona (RM 143–81) reaches. This would help eliminate the small but nevertheless potentially significant survey data gap highlighted by the italicized numbers in Tables 7-2 and 7-3. Surveys for RM 292–243 and RM 143–81 would ideally be conducted every year, but if resources are limited, surveys in alternate years may suffice. Initial results from two or more consecutive years of surveys for the reaches in question might help shed light on an acceptable frequency for future monitoring.

7.7.2.9 Develop linked physical process and biotic response models

The existing model of progressive meander migration (Larsen et al. 2002), and possibly a model of chute cutoff processes, should be linked to models of biotic responses (i.e., metapopulation models similar to that of Moffatt et al. 2005 or a more complex model if sufficient data are available for parameterization) to predict the effects of future management actions on bank swallow habitat and population response.

Ultimately, such linked process-habitat-biotic response models should be used to refine the population viability analysis conducted by CDFG (1992) to improve estimates of population size and colony distribution needed to promote recovery and maintain a viable population.

7.7.3 Potential performance metrics

Based on our understanding of the physical factors affecting bank swallow habitat and the dynamics of bank swallow colonies and populations, we recommend that the following metrics be considered to monitor the health of bank swallows and their habitat along the middle Sacramento River corridor. Continued concurrent monitoring of bank swallows and the metrics would be required to test whether the suggest metrics are indeed useful in this context.

These metrics should be tracked, as appropriate, at various spatial scales:

- The middle river from Redding to Verona (RM 293–81)
- The primary active alluvial reach from Red Bluff to Colusa (RM 243–143)
- Sub-reaches, such as Colusa to Verona (RM 143–81), etc.

- Finer scale sections, such as 5-mile sections (as in Figures 7-13 and 7-14)

7.7.3.1 Physical process and habitat metrics

- Rate of progressive channel meander migration per unit time (average per year), potentially with different targets related to weather cycles (i.e., lower annual rate target during droughts/dry water years and higher targets during wet years)
- Distribution and abundance of vertical banks, by suitability levels (based on Garrison's original HSI model or a modified version of it if more data become available)
- Frequency and duration of flow events above some threshold (25,000 cfs?) during the primary breeding season (April–June) (high values would be *bad*)
- Frequency and duration of flow events above some threshold (25,000 cfs?) during the non-breeding season (August–March) (high values would be generally be *good*, although there would likely be some negative effects associated with more extreme flood events)

7.7.3.2 Biological metrics

- Total number of colonies
- Total number of breeding pairs (continue estimating based on burrow counts, but with periodic checks on percent occupancy of burrows to calibrate/validate estimates)
- Mean colony size and the distribution of colony sizes (as in Figure 7-16)
- Clutch size
- Reproductive success (fledglings per nest, or similar measure)

7.7.4 Key hypotheses and uncertainties

The following hypotheses and uncertainties warrant further study to improve our ability to predict effects of different management actions and natural events on bank swallows along the middle Sacramento River corridor. Key *hypotheses* that need to be tested include:

- Removal or retirement of riprap or other bank protection along banks with suitable nesting sites will lead to colonization in successive breeding seasons—especially at sites that historically supported bank swallow colonies. Removal of riprap in carefully selected areas will renew lateral migration and generate fresh surfaces for nesting in suitable soils, and will alter the hydraulics of the river in the vicinity of the removal site. This hypothesis appears to be supported by observations of bank swallow habitat use after levee removal at RM 233 (Section 7.7.2.4), but further testing of this hypothesis is still needed.
- Increasing progressive meander migration is generally—but not always—good for bank swallows; lateral migration can sometimes lead the river into floodplain deposits that do not contain suitable soils. At present we do not know the degree to which spatial variability in soil suitability influences interannual variability in colony locations and size.
- In the absence of nearby grasslands for bank swallow foraging, agricultural fields serve as an adequate substitute, despite effects of pesticide use. Similarly, oak savannas can provide substitute foraging habitat.
- Regulation of flow from Keswick Dam can reduce detrimental effects of tributary flow during late spring storms that overlap with the swallow breeding season. However, uncertainty in predicting runoff from spring storms makes maintenance of suitably low

flows during the breeding season (e.g., Figure 7-4) problematic, since dam releases need to be matched with expected tributary inputs in order to effectively manage river stage in the reaches supporting bank swallow colonies. Similarly, the hypothesis that winter flows can and should be managed to benefit bank swallow habitat needs to be tested. More study would be needed to determine how often winter flows might need to be augmented to renew suitable bank nesting habitat (e.g., would this only occur during multiyear drought periods?).

- Maintenance of a surplus of suitable nesting habitat (i.e., banks with high HSI values) is essential. We do not know the precise threshold of unused suitable habitat that is required for recovery and maintenance of a viable population, but it is estimated to be around 40-60% unused habitat (B. Garrison, pers. comm.., 2006).

7.7.4.1 Ideas for future studies to reduce uncertainties

- Reduce uncertainty related to desired level of habitat occupancy
  - Can we accurately predict habitat suitability (i.e., are unoccupied vertical banks actually suitable nesting sites, or are they unsuitable for some unknown reason)? More detailed mapping of soil texture might allow more accurate prediction of habitat suitability.
  - What should be the desired level of site occupancy in a “healthy” system with a viable natural population or metapopulation of bank swallows? How much surplus of suitable nesting habitat is required?
  - How do colonial nesting, site fidelity, and density-dependent factors interact to affect site occupancy levels and the ability of the population or metapopulation to expand when additional nesting habitat becomes available? How do these factors interact with density-independent and stochastic factors such as high flows and bank erosion? Does the theoretical work of Matthiopoulos et al. (2005) help us understand the observed site occupancy levels along the Sacramento River?
  - Continue banding research to determine population movements, population dynamics, and colony site fidelity.
- Reduce uncertainty related to the relative value of different habitats as foraging areas
  - Review available data or conduct new studies to test whether food supply is a likely limiting factor for bank swallows along the Sacramento River. If the food limitation hypothesis is not rejected, consider addressing the following questions:
    - What are the relative levels of prey available in different habitats?
    - How many acres of grassland (or other types of foraging habitat) are needed to support an average colony?
    - How close does foraging habitat have to be to the colony to support optimal population levels and productivity?
    - How do land use activities (such as use of pesticides or herbicides) affect prey abundance?
    - How important is foraging over water compared to foraging over land? Is there a feasible way to manage aquatic habitats to increase production of aquatic insects (i.e., increase emergence of winged adult insects)?
- Reduce uncertainty related to the influence of nest ectoparasites on reproductive success in the Sacramento Valley population
  - What are the diversity, relative abundance and distribution of ectoparasites among and within colonies along the river?

  - Which ectoparasites have the most impact on nestling growth and survival?
  - At what level of ectoparasite loading do negative impacts become evident?
  - Are there environmental factors that are strongly correlated with ectoparasite loads (e.g., bank age or time since last erosion event)? If so, can any of these factors be significantly influenced by feasible management actions?
- Reduce uncertainty regarding importance of winter habitat and migration on population dynamics of Sacramento Valley bank swallows
  - Where are the main centers of winter habitat for this population? What is the current quantity and quality of winter habitat? What are the current threats to and the potential for restoring (if impaired) such habitat?
  - What are the primary migratory routes for birds in this population? What is the current quantity and quality of key migratory habitat (stopover sites for roosting and foraging)? What are the current threats to and the potential for restoring (if impaired) such habitat?
  - Can we use trace mineral and isotope studies of feathers to help identify important molting areas within the winter habitat range, as has been done for Old World bank swallows (e.g., Szep et al. 2003).
  - To what degree are bank swallows limited by factors on the breeding grounds vs. factors operating in wintering or migratory areas?

7.8 References

Arny, S. A. 1952. Taxonomic status of the bank swallow of North America. The Condor 54: 356-357.

Blem, C. R. 1979. Predation of black rat snakes on a bank swallow colony. The Wilson Bulletin 91: 135-137.

Brown, C.R. and M. B. Brown. 1996. Coloniality in the cliff swallow. The effect of group size on social behavior. University of Chicago Press, Chicago, Illinois.

Buechner, M. 1992. Preliminary population viability analysis for bank swallows (*Riparia riparia*) on the Sacramento River, California: a computer simulation analysis incorporating environmental stochasticity. California Department of Fish and Game, Nongame Bird and Mammal Section.

Buer, K. 1994. Sacramento River bank erosion investigation memorandum progress report. Internal memorandum to R. Scott and L. Brown from K. Buer, Chief, Geology Section, California Department of Water Resources, Northern District, Red Bluff.

Buer, K. 1995. Sacramento River gravel study: Keswick Dam to Cottonwood Creek. California Department of Water Resources, Northern District, Red Bluff.

CDFG (California Department of Fish and Game). 1992. Recovery plan: bank swallow. Report No. 93.02. CDFG, Nongame Bird and Mammal Section, Wildlife Management Division, Sacramento.

CDFG. 1997. Annual report on the status of California state listed threatened and endangered animals and plants. Fish and Game Commission, Sacramento.

Clegg, M. 1977. Sand martins feeding on the ground. British Birds 70: 361.

Constantine, C., T. Dunne, and M. Singer. Unpublished. Controls on spatial differences in meander migration rates in a large gravel-bed river. Submitted to Earth Surface Processes and Landforms.

Cramp, S. 1988. The birds of the western Palearctic. Volume 5: tyrant flycatchers to thrushes. Oxford University Press, Oxford, England.

Drake, V. A., and R. A. Farrow. 1988. The influence of atmospheric structure and motions on insect migration. Annual Review of Entomology 33: 183-210.

Ehrlich, P. R., D. S. Dobkin, and D. Wheye. 1988. The birder's handbook: A field guide to the natural history of North American birds. Simon and Schuster, Inc., New York.

Freer, V. M. 1977. Colony structure and function in the bank swallow, *Riparia riparia* L. Doctoral dissertation. State University of New York, Binghamton.

Gardali, T., S. L. Small, N. Nur, G. R. Geupel, G. Ballard, and A. L. Holmes. 2005. Monitoring songbirds in the Sacramento Valley (1993-2003): population health, management information, and restoration evaluation. PRBO Contribution No. 1233 Prepared by Point Reyes Bird Observatory, Stinson Beach, California for The Nature Conservancy, U.S. Fish and Wildlife Service, and River Partners.

Garrison, B. A. 1989. Habitat suitability index models: bank swallow. U.S. Fish and Wildlife Service, Sacramento, California.

Garrison, B. A. 1991. Evaluation of experimental nesting habitat and selected aspects of bank swallow biology on the Sacramento River, California, 1988 to 1990. U.S. Fish and Wildlife Service, Sacramento, California.

Garrison, B. A. 1998. Bank swallow (*Riparia riparia*). In California Partners in Flight Riparian Bird Conservation Plan: a strategy for reversing the decline of riparian-associated birds in California. California Partners in Flight, Point Reyes Bird Observatory, Stinson Beach, California. http://www.prbo.org/calpif/htmldocs/species/riparian/bank_swallow_acct2.html

Garrison, B. A. 1999. Bank swallow (*Riparia riparia*). No. 414. *In* A. Poole and F. Gill, editors. The birds of North America. The Academy of Natural Sciences, Philadelphia, Pennsylvania and The American Ornithologists' Union, Washington, D. C.

Garrison, B. A., J. M. Humphrey, and S. A. Laymon. 1987. Bank swallow distribution and nesting ecology on the Sacramento River, California. Western Birds 18: 71-76.

Garrison, B., and R. McKernan. 1994. Bank swallow. Pages 208-209 *in* C. G. Thelander and M. Crabtree, editors. Life on the edge: a guide to California's endangered natural resources: wildlife. Biosystems Books, Santa Cruz, California.

Golet, G. H., D. L. Brown, E. E. Crone, G. R. Geupel, S. E. Greco, K. D. Holl, D. E. Jukkola, G. M. Kondolf, E. W. Larsen, F. K. Ligon, R. A. Luster, M. P. Marchetti, N. Nur, B. K. Orr, D. R. Peterson, M. E. Power, W. E. Rainey, M. D. Roberts, J. G. Silveira, S. L. Small, J. C. Vick, D. S. Wilson, and D. M. Wood. 2003. Using science to evaluate restoration efforts and ecosystem health on the Sacramento River Project, California. Pages 368-385 *in* P. M. Faber, editor. California riparian systems: processes and floodplain management, ecology, and restoration. 2001. Riparian habitat and floodplain conference proceedings. Riparian Habitat Joint Venture, Sacramento, California.

Greco, S. E., A. K. Fremier, E. W. Larsen, and R. E. Plant. Unpublished (2006a). Tracking surficial patterns of floodplain chronology on a large meandering river: analysis of land production rates and riparian vegetation distribution over land age gradients. Submitted to Ecography.

Greco, S. E., E. H. Girvetz, E. W. Larsen, J. P. Mann, J. L. Tuil, and C. Lowney. Unpublished (2006b). A method to model a relative elevation topographic surface of a large alluvial river floodplain and riparian ecological application. Submitted to Landscape Research.

Gross, A. O. 1942. Bank swallow. Pages 400-424 *in* A. C. Bent, editor. Life histories of North American flycatchers, larks, swallows, and their allies. U.S. National Museum Bulletin 179. Smithsonian Institution, Washington, D. C.

Haas, G. E., T. Rumfelt, and N. Wilson. 1980. Fleas (*Siphonaptera*) from nests and burrows of the bank swallow (*Riparia riparia*) in Alaska. Northwest Science 54: 210-215.

Hickman, G. R. 1979. Nesting ecology of bank swallows in interior Alaska. Master's thesis. University of Alaska, Fairbanks.

Hight, R. C. 2000. Annual report on the status of the threatened bank swallow. California Department of Fish and Game.

Hjertaas, D. G. 1984. Colony site selection in bank swallows. Master's thesis. University of Saskatchewan, Saskatoon.

Hobson, K. A., and S. G. Sealy. 1987. Foraging, scavenging, and other behavior of swallows on the ground. The Wilson Bulletin 99: 111-116.

Hoogland, J. L., and P. W. Sherman. 1976. Advantages and disadvantages of bank swallow (*Riparia riparia*) coloniality. Ecological Monographs 46: 33-58.

Humphrey, J. M., and B. A. Garrison. 1987. The status of bank swallow populations on the Sacramento River. Final report. California Department of Fish and Game, Wildlife Management Division, Sacramento.

Klinesteker, S. 1998. Draft Sacramento River meander belt—future erosion study. Memorandum to K. Buer. California Department of Water Resources, Northern District, Red Bluff.

Kondolf, G. M., T. Griggs, E. W. Larsen, S. McBain, M. Tompkins, J. G. Williams, and J. Vick. 2000. Flow regime requirements for habitat restoration along the Sacramento River between Colusa and Red Bluff. CALFED Bay Delta Program, Integrated Storage Investigation, Sacramento, California.

Kuhnen, K. 1985. On pair-formation in the sand martin, *Riparia riparia.* Journal of Ornithology 126: 1-13.

Larsen, E. W., and S. E. Greco. 2002. Modeling channel management impacts on river migration: a case study of Woodson Bridge State Recreation Area, Sacramento River, California, USA. Environmental Management 30: 209-224.

Larsen, E. W., A. K. Fremier, and S. E. Greco. Unpublished. Cumulative effective stream power and river channel migration on the Sacramento River, California, USA. Submitted to Environmental Management.

Laymon, S. A., B. A. Garrison, and J. M. Humphrey. 1988. Historic and current status of the bank swallow in California, 1987. Administrative Report 88-2. California Department of Fish and Game, Wildlife Management Division, Sacramento.

Matthiopoulos, J., J. Harwood, and L. Thomas. 2005. Metapopulation consequences of site fidelity for colonially breeding mammals and birds. Journal of Animal Ecology 74: 716-727.

Mayr, E., and J. C. Greenway, Jr. 1960. Check-list of birds of the world. Volume 9. Museum of Comparative Zoology, Cambridge, Massachusetts.

Mead, C. J. 1979. Colony fidelity and interchange in the sand martin. Bird Study 26: 99-106.

Micheli, E. R., and E. W. Larsen. In preparation. River channel cutoff dynamics, Sacramento River, California, USA.

Micheli, E. R., J. W. Kirchner, and E. W. Larsen. 2004. Quantifying the effect of riparian forest versus agricultural vegetation on river meander rates, central Sacramento River, California, USA. River Research and Applications 20: 537-548.

Micheli, E. R., and E. W. Larsen. In preparation. River channel cutoff dynamics, Sacramento River, California, USA.

Moffatt, K. C., E. E. Crone, K. D. Holl, R. W. Schlorff, and B. A. Garrison. 2005. Importance of hydrologic and landscape heterogeneity for restoring bank swallow (*Riparia riparia*) colonies along the Sacramento River, California. Restoration Ecology 13: 391-402.

Oberholser, H. 1974. The bird life of Texas. University of Texas Press, Austin.

Petersen, A. J. 1955. The breeding cycle in the bank swallow. The Wilson Bulletin 67: 235-286.

Petersen. C. 2005. Beachfront property: bank swallows dig in at Fort Funston. BayNature 5(3): 30-32. http://www.baynature.com/2005julysept/fortfunston.html

Phillips, A.R. 1986. The known birds of North and Middle America: distributions and variation, migrations, changes, hybrids, etc. Part I, Hirundinidae to Mimidae, Certhiidae. Denver, Colorado.

Remsen, J. V., Jr. 1978. Bird species of special concern in California: an annotated list of declining or vulnerable bird species. Administrative Report No. 78-1. California Department of Fish and Game, Wildlife Management Branch, Sacramento.

Schlorff, R. W. 1997. Monitoring bank swallow populations on the Sacramento River: a decade of decline. Transactions of the Western Section of the Wildlife Society 33: 40-48.

Schlorff, R. W. 1998-2004. Bank swallow population surveys, Sacramento River. Internal memoranda for files. California Department of Fish and Game, Habitat Conservation Division, Resource Assessment Division, Sacramento.

Sibley, D. A. 2000. The Sibley guide to birds. Alfred A. Knopf, New York.

Sieber, O. J. 1980. Causal and functional aspects of brood distribution in sand martins (*Riparia riparia* L.). Zeitschrift fur Tierpsychologie 52: 19-56.

Small, A. 1994. California birds: their status and distribution. Ibis Publishing Company, Vista, California.

Snowden, V. L. 2002. Hydrologic alterations to the Sacramento River and the effects on cottonwood seedlings. Master's thesis. California State University, Chico.

Stoner, D. 1936. Studies on the bank swallow *Riparia riparia riparia* (Linnaeus) in the Oneida Lake region. Roosevelt Wild Life Bulletin 9: 122-233.

Szep, T., and A. P. Moller. 1999. Cost of parasitism and host immune defense in the sand martin *Riparia riparia*: a role for parentoffspring conflict? Oecologia 119: 9-15.

Szep, T., A. P. Moller, J. Vallner, B. Kovacs, and D. Norman. 2003. Use of trace elements in feathers of sand martin *Riparia riparia* for identifying moult areas. Journal of Avian Biology 34: 307-320.

Turner, A., and C. Rose. 1989. Swallows and martins: an identification guide and handbook. Houghton Mifflin, Boston, Massachusetts.

USFWS (U.S. Fish and Wildlife Service). 1980. Habitat Evaluation Procedures (HEP). ESM 102. USFWS, Division of Ecological Services.

Zeiner, D. C., W. F. Laudenslayer Jr., K. E. Mayer, and M. White, editors. 1990. California's wildlife. Volume II. Birds. California Statewide Habitat Relationships System. California Department of Fish and Game.

Figures

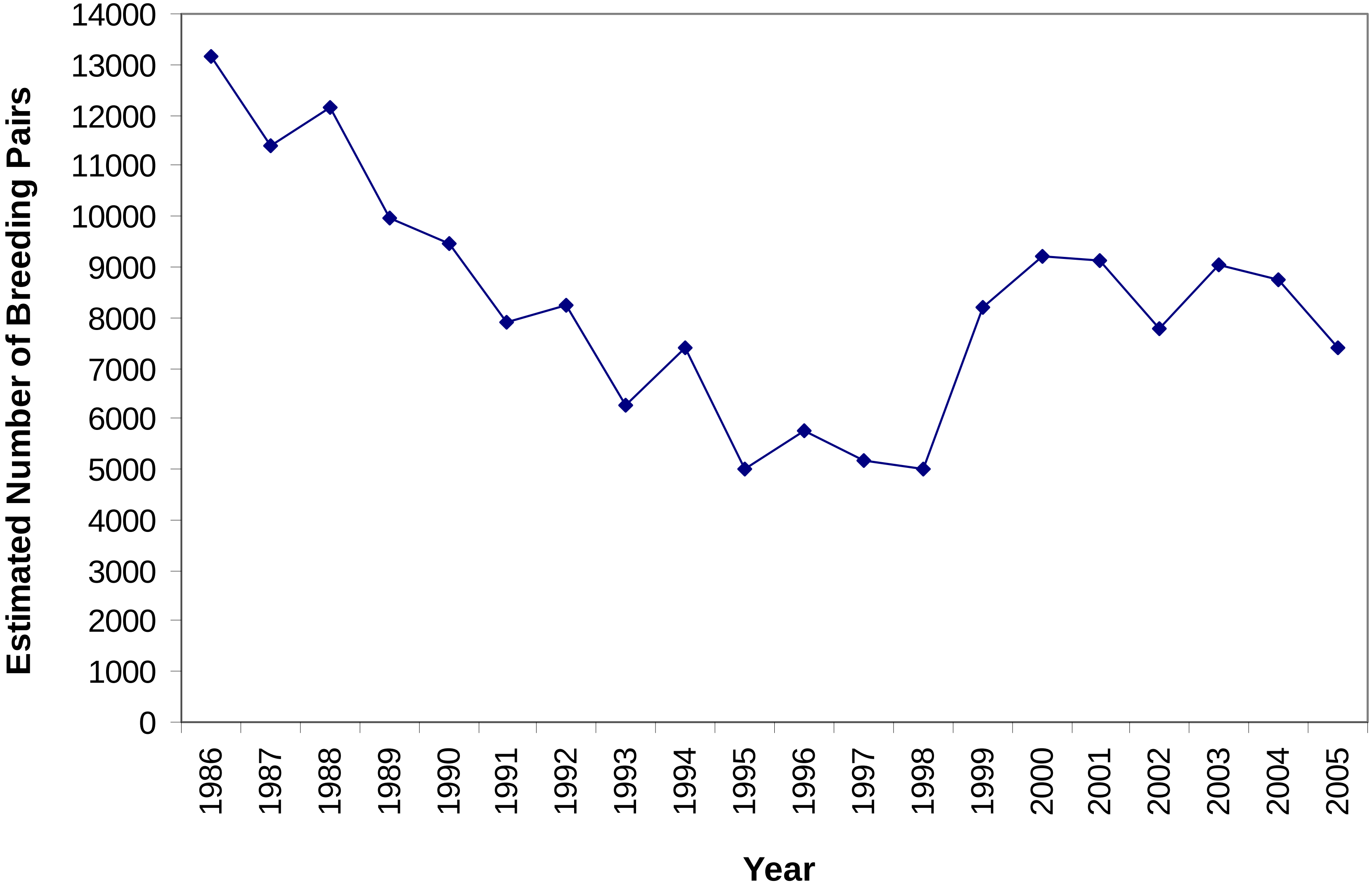
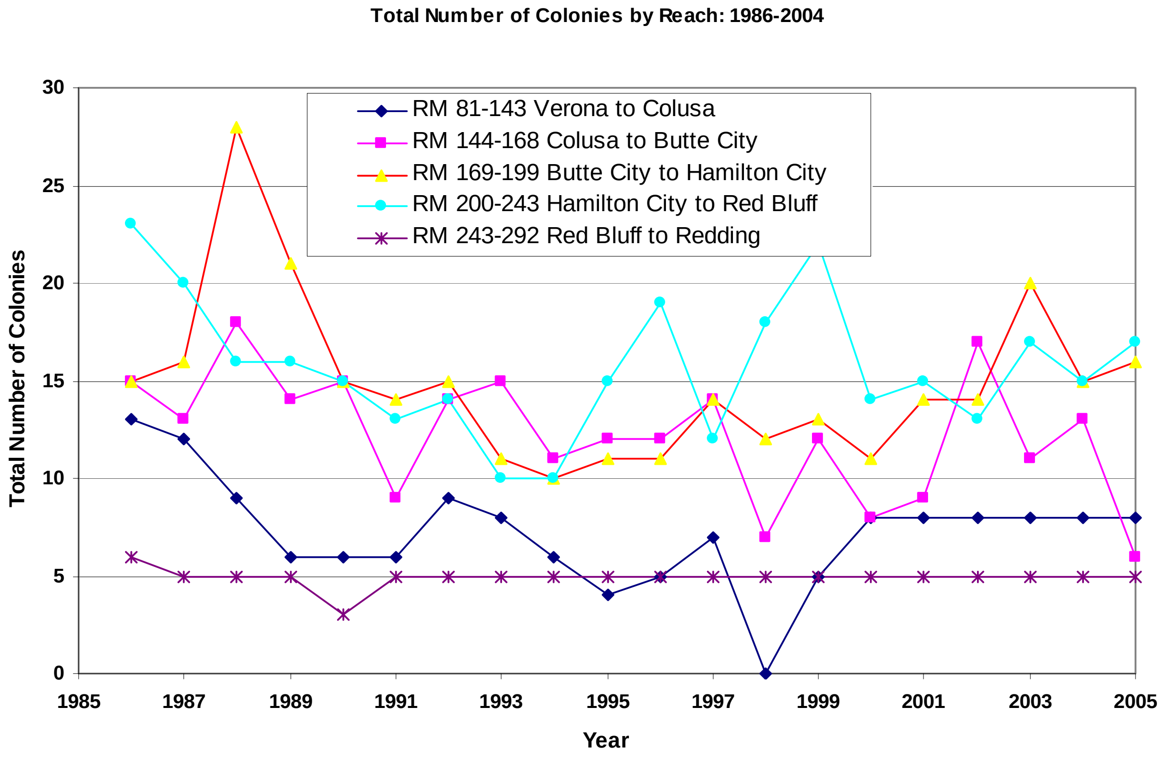


Figure 7-1. Bank swallow population along the Sacramento River corridor as a function of time. Note the steep, nearly continuous decline from 1986 to 1998, followed by partial rebound and stabilization at roughly 8,000 to 9,000 pairs in recent years. Source: Hight 2000, Schlorff 1997, 1998, 1999, 200, 2001, 2002, 2003, 2004, 2005.



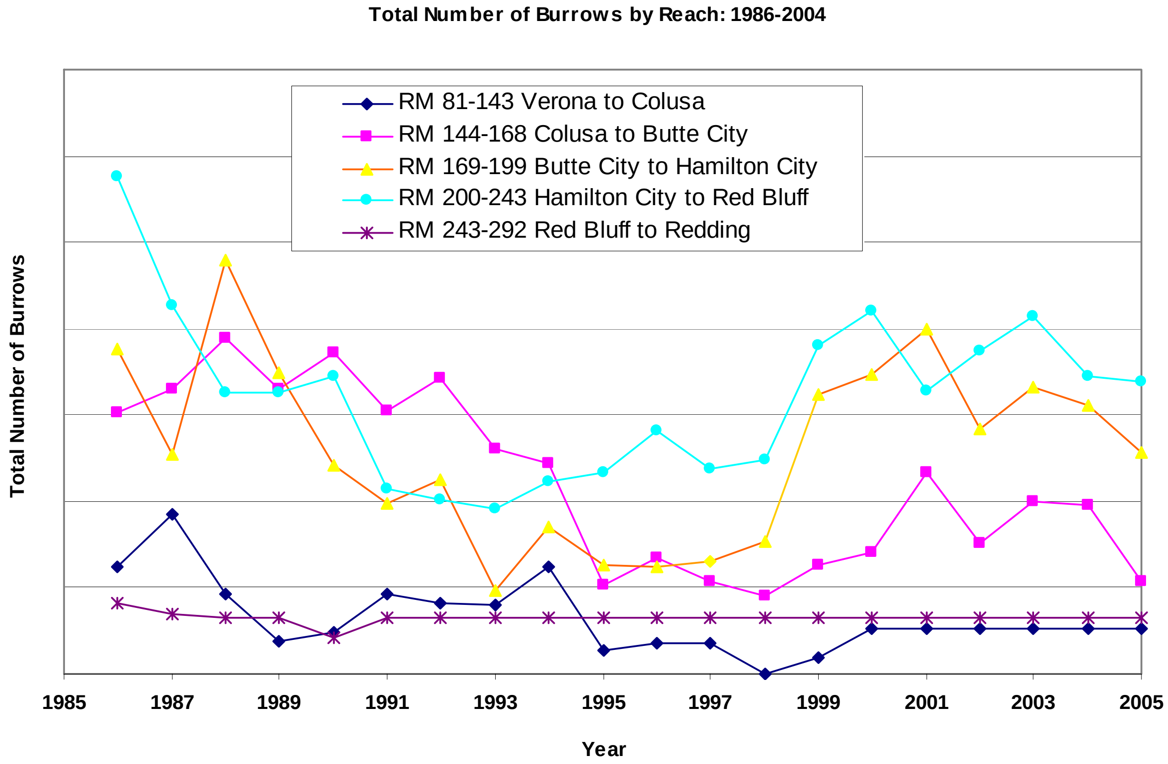


Figure 7-2. Number of bank swallow colonies (top) and burrows (bottom) from 1986-2005 for each of the five CDFG reaches. Source: Hight 2000, Schlorff 1997, 1998, 1999, 2000, 2001, 2002, 2003, 2004, 2005.

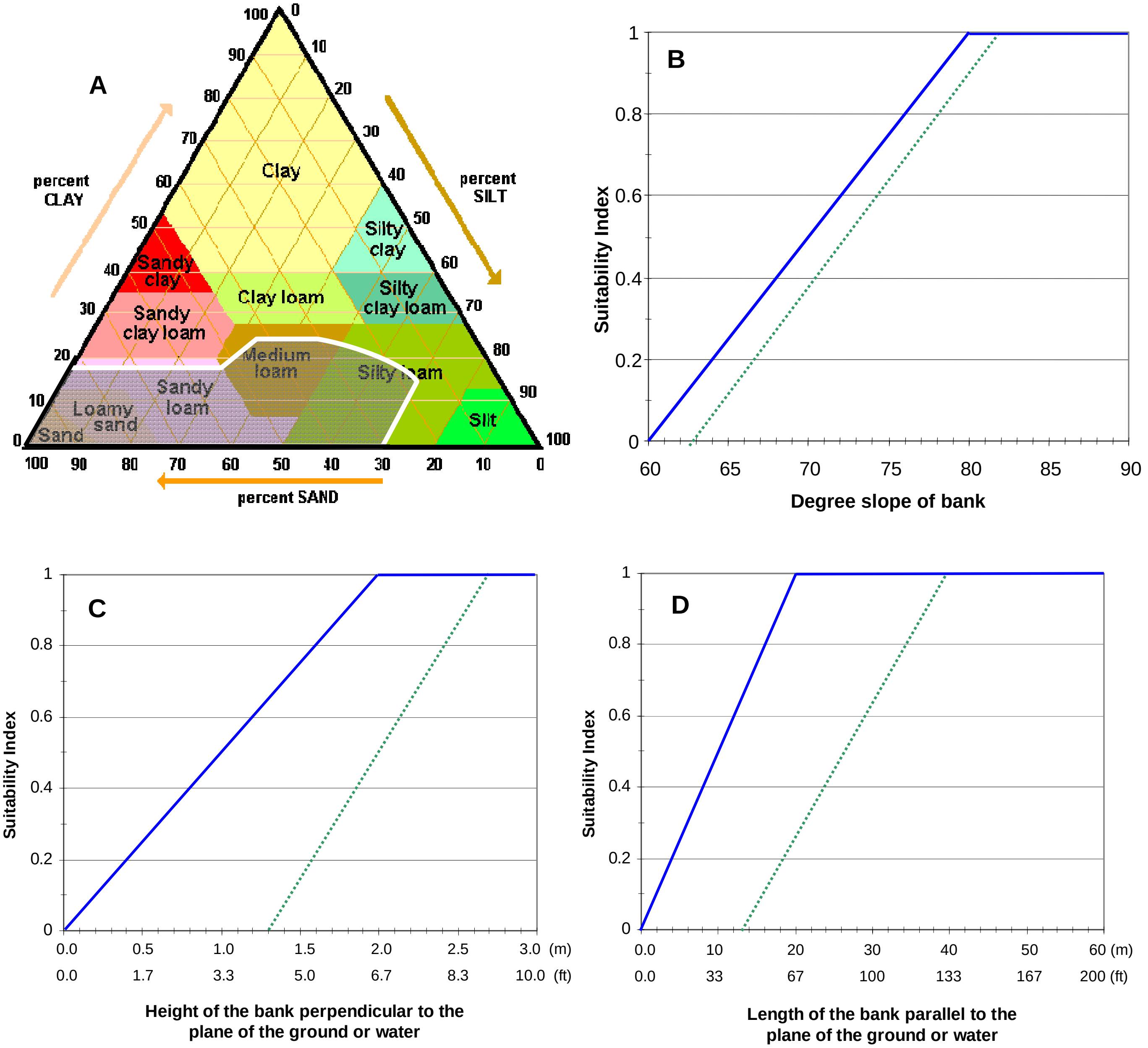


Figure 7-3a, b, c, d. Habitat suitability index (HSI) relationships for four key variables affecting bank swallow breeding habitat, based on the HSI model developed by Garrison (1989). Suitability values can range from unsuitable (0) to fully suitable (1). Solid lines indicate relationship proposed by Garrison (1989). Dashed lines indicate potential adjustments based on new interpretations (this report) of Sacramento River colony data collected by Garrison and others (1987). A. Soil textures suitable for nesting are represented by the shaded region in the lower left-center portion of the soil texture triangle (these soil types have suitability index = 1, all others = 0). B. Relationship between bank slope and suitability (near vertical banks have suitability = 1). C. Relationship of bank height above base flow water surface elevation during the nesting season to habitat suitabilty (taller banks have greater suitability). D. Habitat suitability increases with length of freshly eroded bank available.

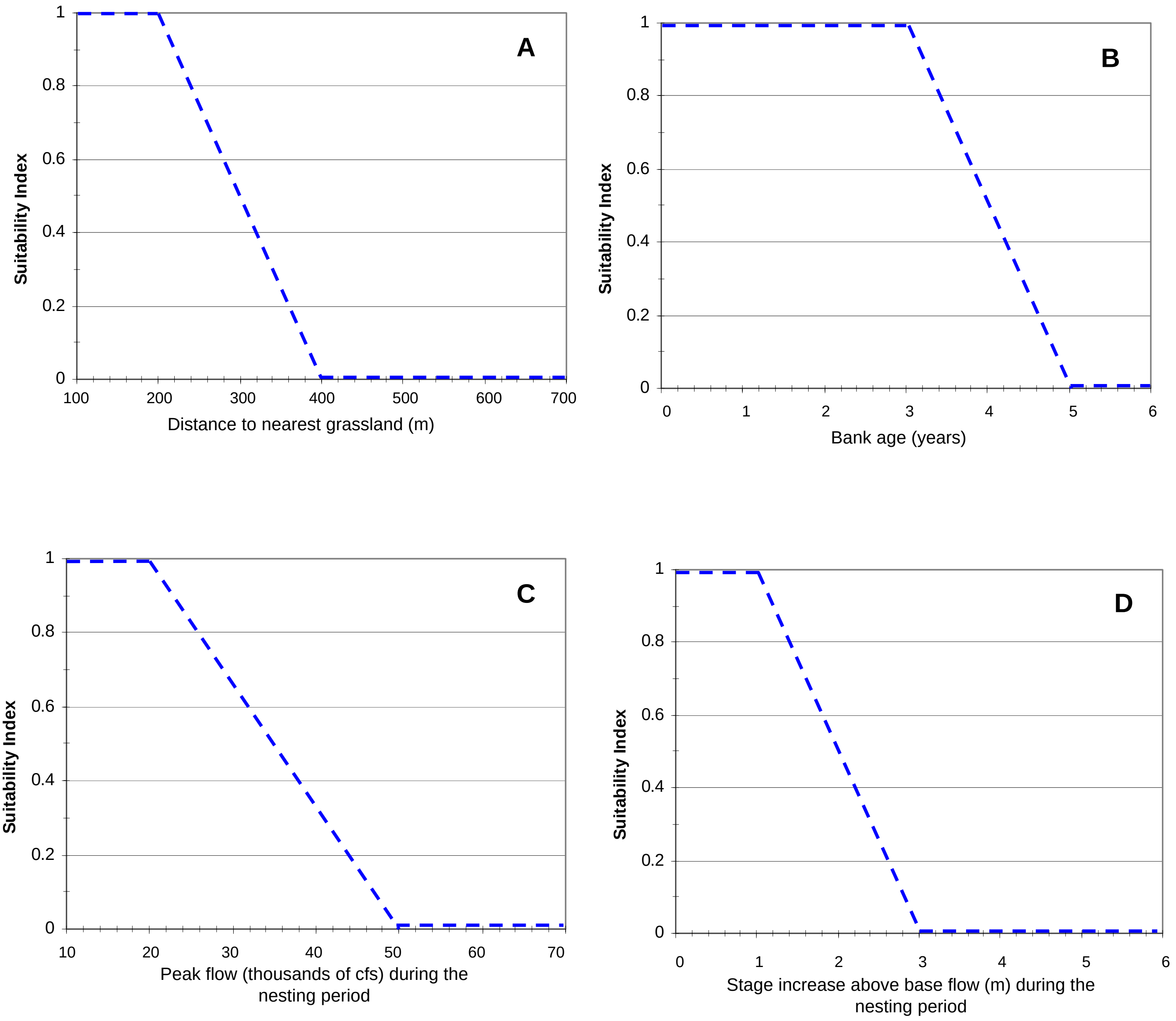


Figure 7-4a, b, c, d. Hypothesized relationships between physical habitat variables and suitability for bank swallow nesting. A. Distance to nearest grassland area (good foraging habitat) affects habitat suitability (based on data from Moffat et al 2005). B. Bank age (time since last major erosion event at that bank) affects suitability; banks greater than 3 years old tend to rapidly become more susceptible to predation (based on personal communications with B. Garrison and R. Schlorff, 2005). C. Peak flows in excess of 20,000 or 25,000 cfs during the nesting period have the potential to cause bank erosion, which if severe enough can result in partial or complete loss of colonies (see text for details). D. Large increases in river stage (water surface elevation) during the nesting season have the potential to cause direct mortality of eggs, nestlings, or even adults due to nest inundation and subsequent drowning or burrow collapse (personal communications with B. Garrison and R. Schlorff, 2005).

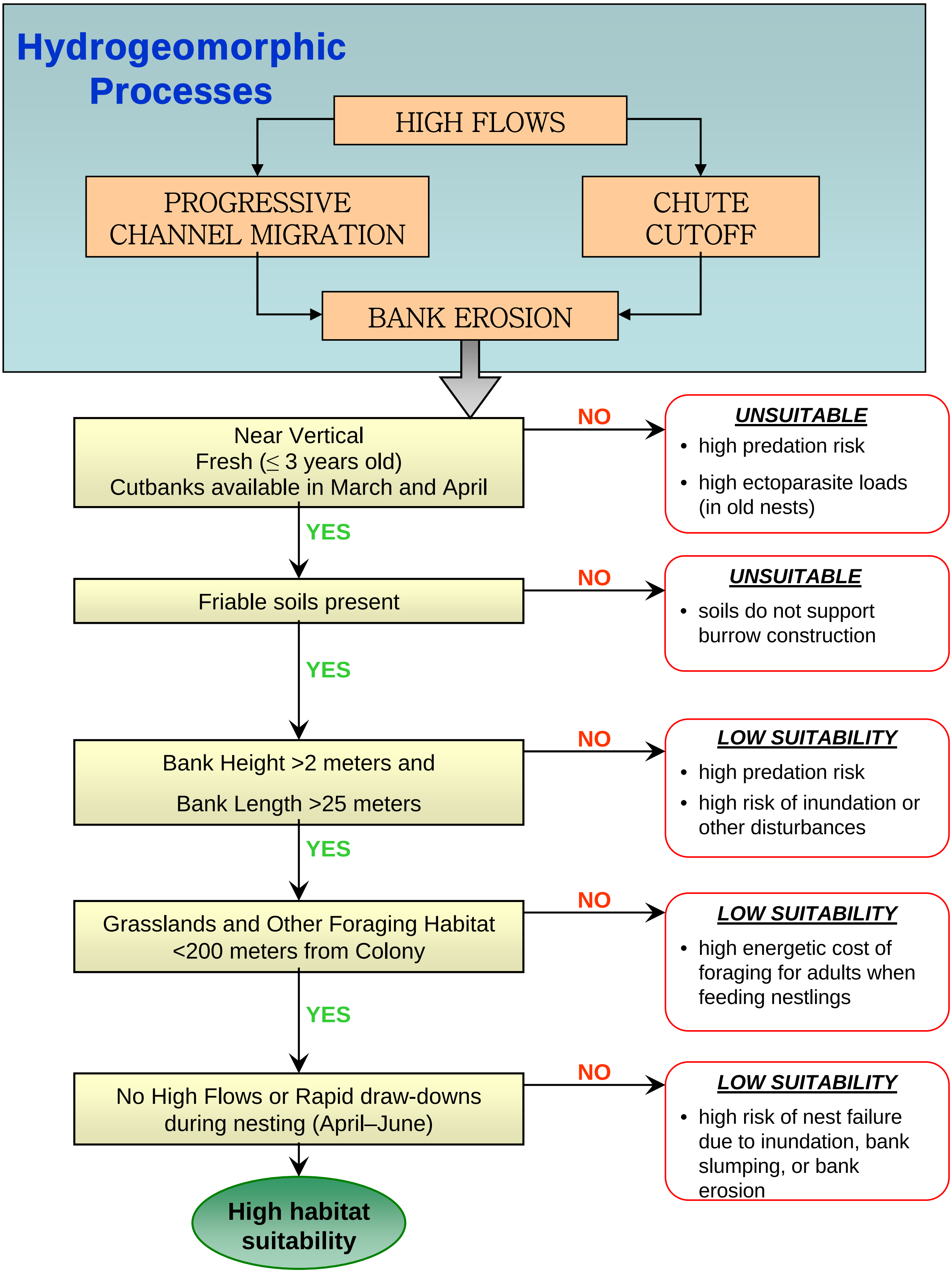


Figure 7-5. General conceptual model of influences of hydrogeomorphic processes and physical habitat conditions on habitat suitability for bank swallow nesting in riverine systems.

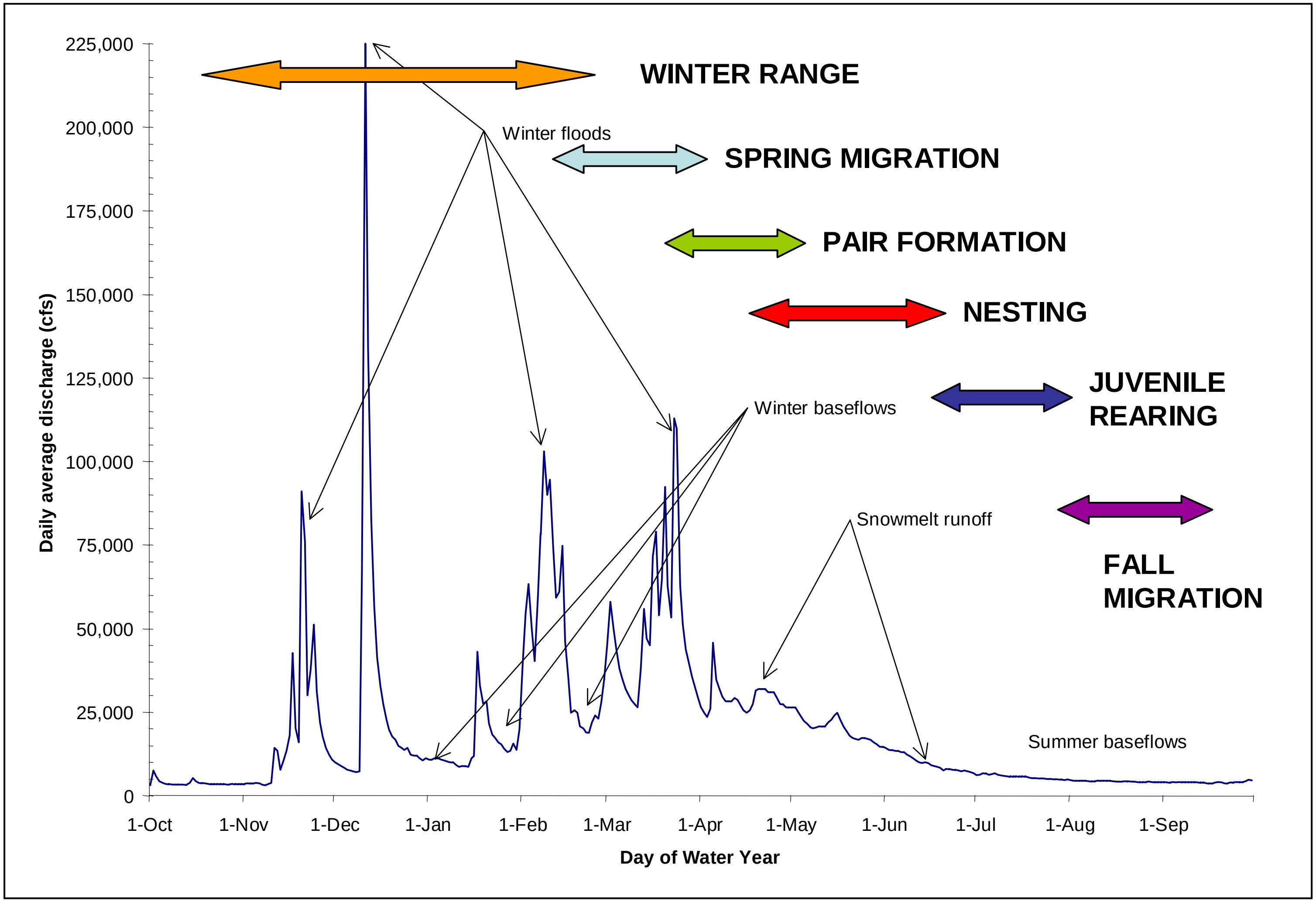


Figure 7-6. Representative annual hydrograph from the pre-Shasta period showing key hydrograph components relative to bank swallow life history stages: overwintering (orange), spring migration (light blue), pair bonding (green), nesting (red), juvenile rearing and local dispersal (dark blue), and fall migration (purple) periods. Hydrograph data are from the Bend Bridge Gage (near Red Bluff) for Water Year 1938 (CALFED 2000).

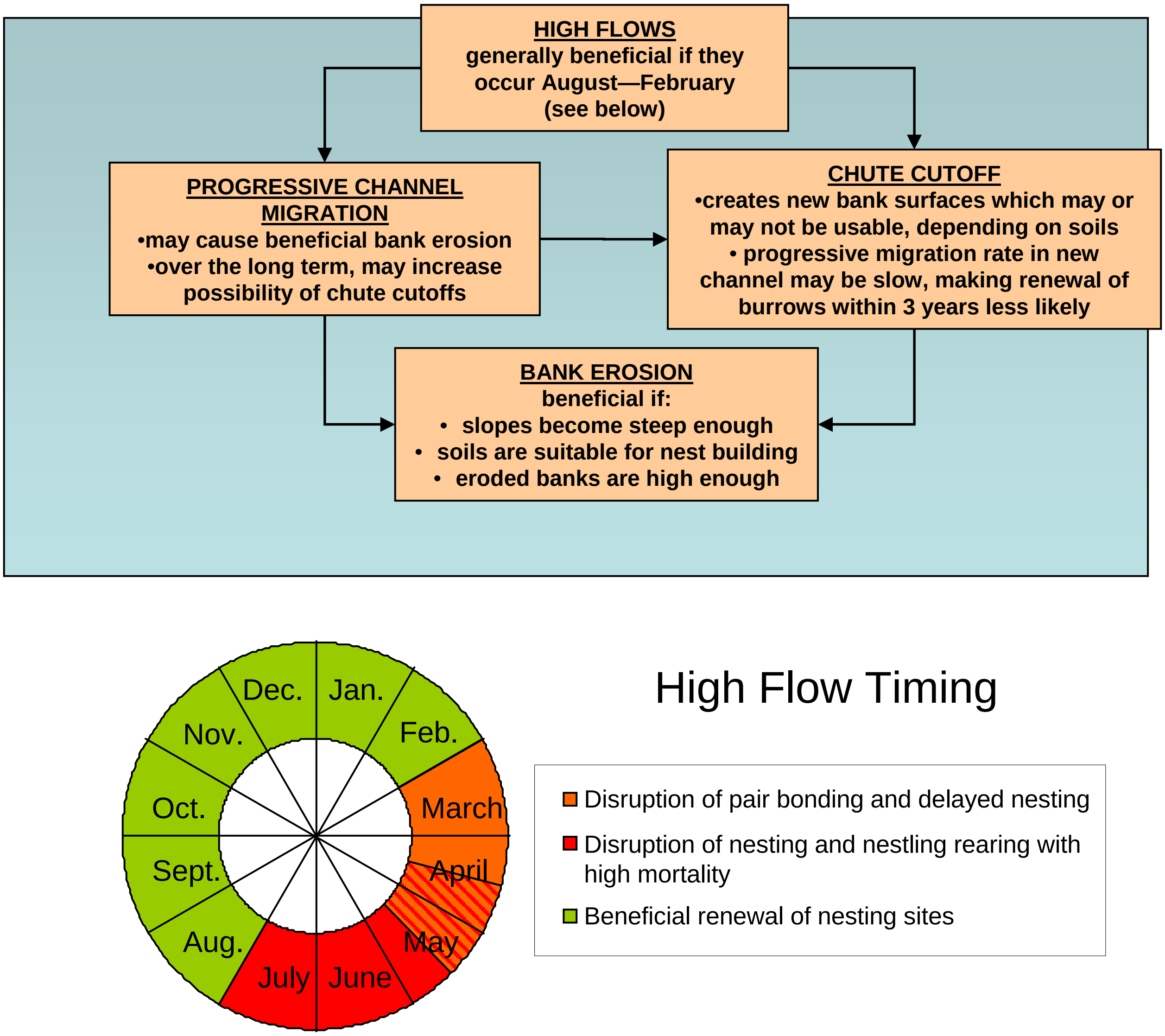


Figure 7-7. Conceptual diagram illustrating how timing of high flow events interacts with bank swallow phenology (life history timing) to determine likely effects on bank swallow populations in the Sacramento River valley. High flows occurring in August through February are generally beneficial to nesting sites, causing erosion that removes old nests (which eventually become infested with fleas) and keeps banks steep—and thus inaccessible to predators. High flows during March, April, and early May can cause nest inundation and bank collapse and thus disrupt pair bonding, leading to delayed nesting and potentially lower breeding success. Nest inundation and bank collapse due to high flows occurring in late April through July can disrupt nesting and nestling rearing and lead to high bank swallow mortality. Chute cutoff can quickly create long new stretches of suitable bank swallow habitat (if bank soils are suitable), but may strand existing habitat (in the old main channel), making further renewal there unlikely. Moreover, new chute cutoffs may have low migration rates (due, for example, to reduced sinuosity).

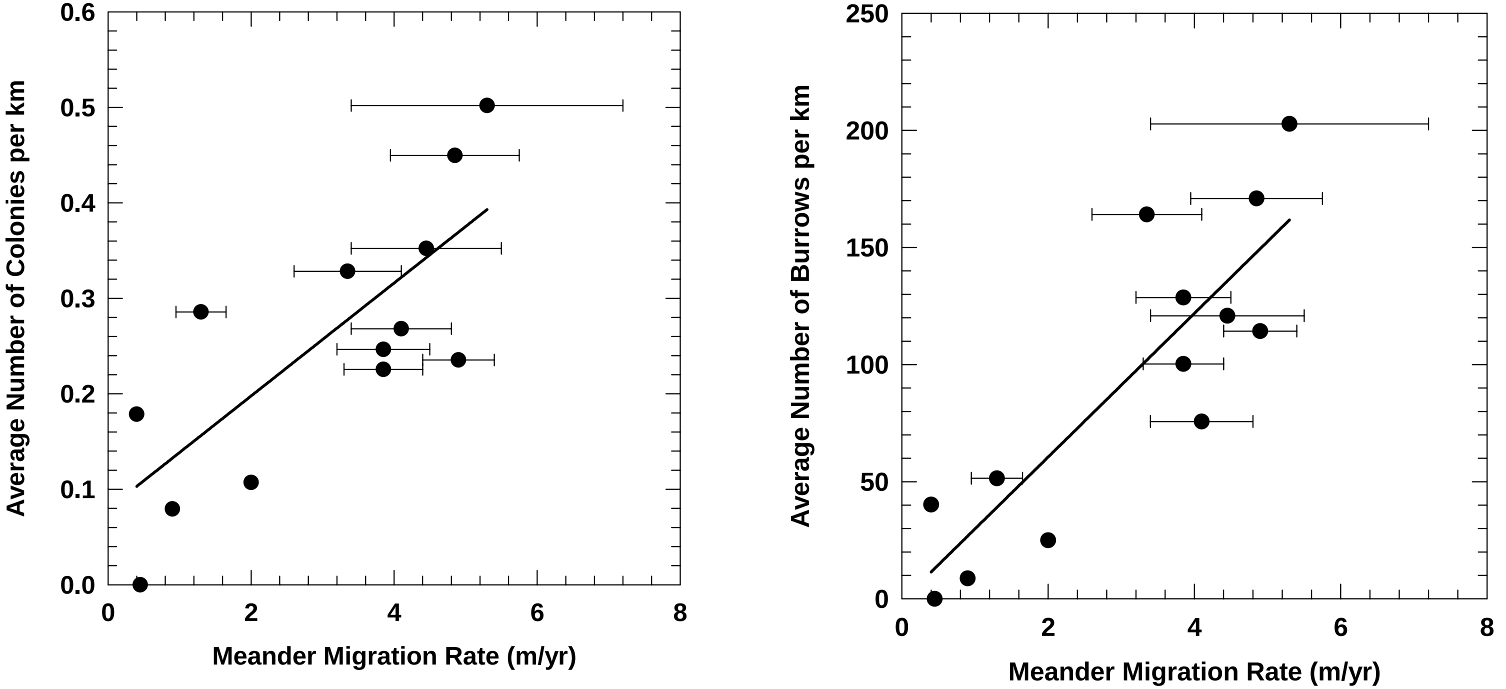


Figure 7-8. Average number of bank swallow colonies (left plot) and bank swallow burrows (right plot) per km of 1997 centerline channel length plotted against average meander migration rates for the Sacramento River. The densities of colonies and burrows both increase with increasing meander migration rates. Banks swallow data are averaged by reach for 15 active and stable reaches (as defined by Constantine et al. *in press*) using data from the 1998-2004 colony surveys (Schlorff, 1997, 1998, 1999, 2000, 2001, 2002, 2003, 2004). Meander migration rates are averaged over the post-dam interval (1946-1997) for each of the 15 reaches (Constantine et al. *in press*).

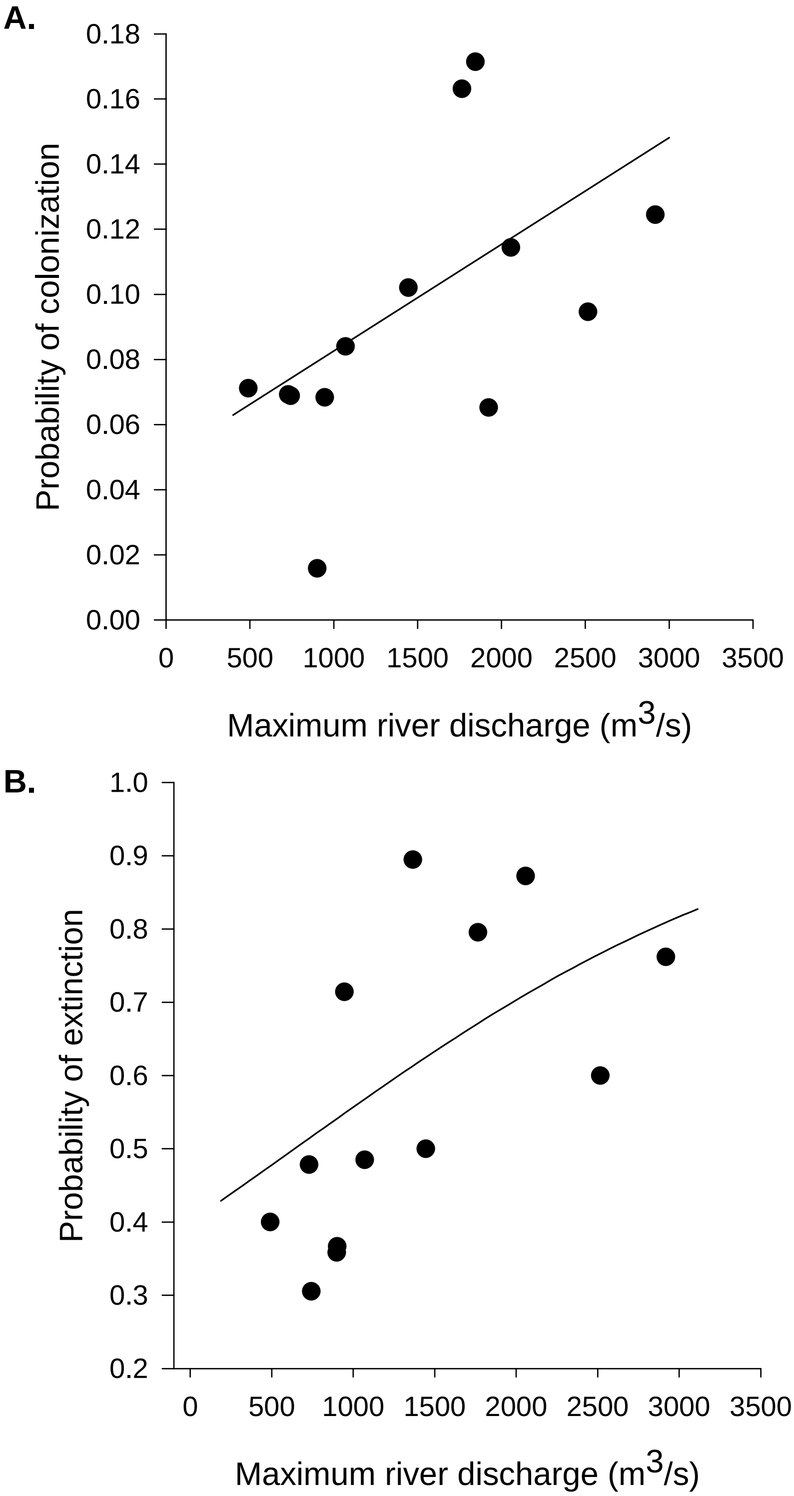


Figure 7-9. Relationship between maximum river discharge and (A) colonization rate and (B) extinction rate (Source: Figure 4 in Moffat et al. 2005). Data are yearly colonization and extinction probabilities from 1986-1992 and 1996-2003, calculated across all sites (i.e., the "temporal" analyses described in Moffat et al. 2005). Note: 1 cfs = 0.02832 m3/s, and 1 m3/s = 35.31 cfs.

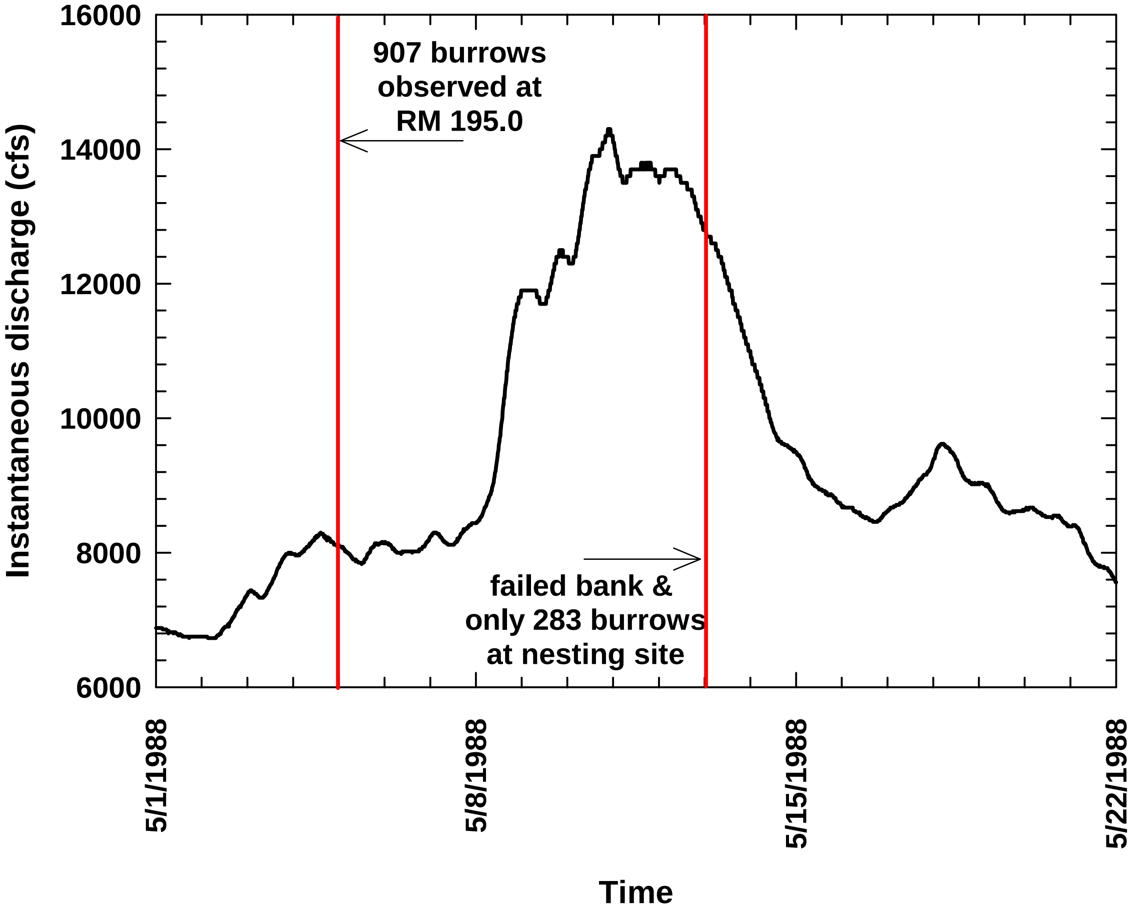


Figure 7-10. Sequential observations (B. Garrison pers. comm. 2005) of abundant bank swallows and a partial colony failure due to bank collapse at RM 195 (red lines) bracketing a period in which flow (based on 15 minute data) peaked at 14,000 cfs at the Butte City gauge downstream at RM 169.

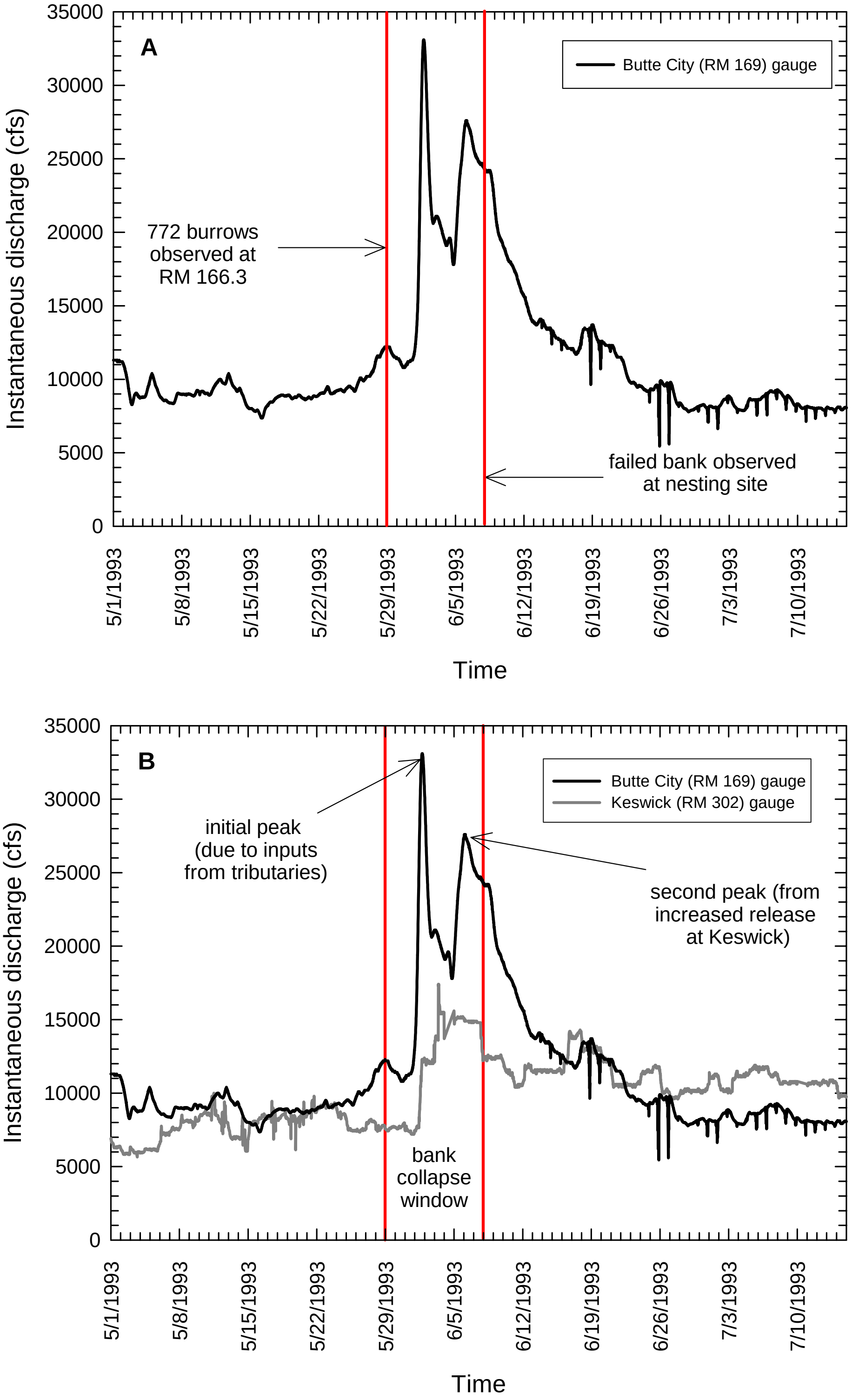


Figure 7-11. A: 15 minute discharge data at Butte City (RM 169) and sequential observations (B. Garrison, pers. comm. 2005) of abundant bank swallows and a collapsed bank at RM 166.3 (red lines) bracketing two flow peaks >25,000 cfs. B: The fact that flow at Keswick (gray line) did not peak until after the first peak at Butte City (black line) suggests that the first peak was probably due to inputs from unregulated tributaries and that the second peak was probably a delayed response to Keswick flow.

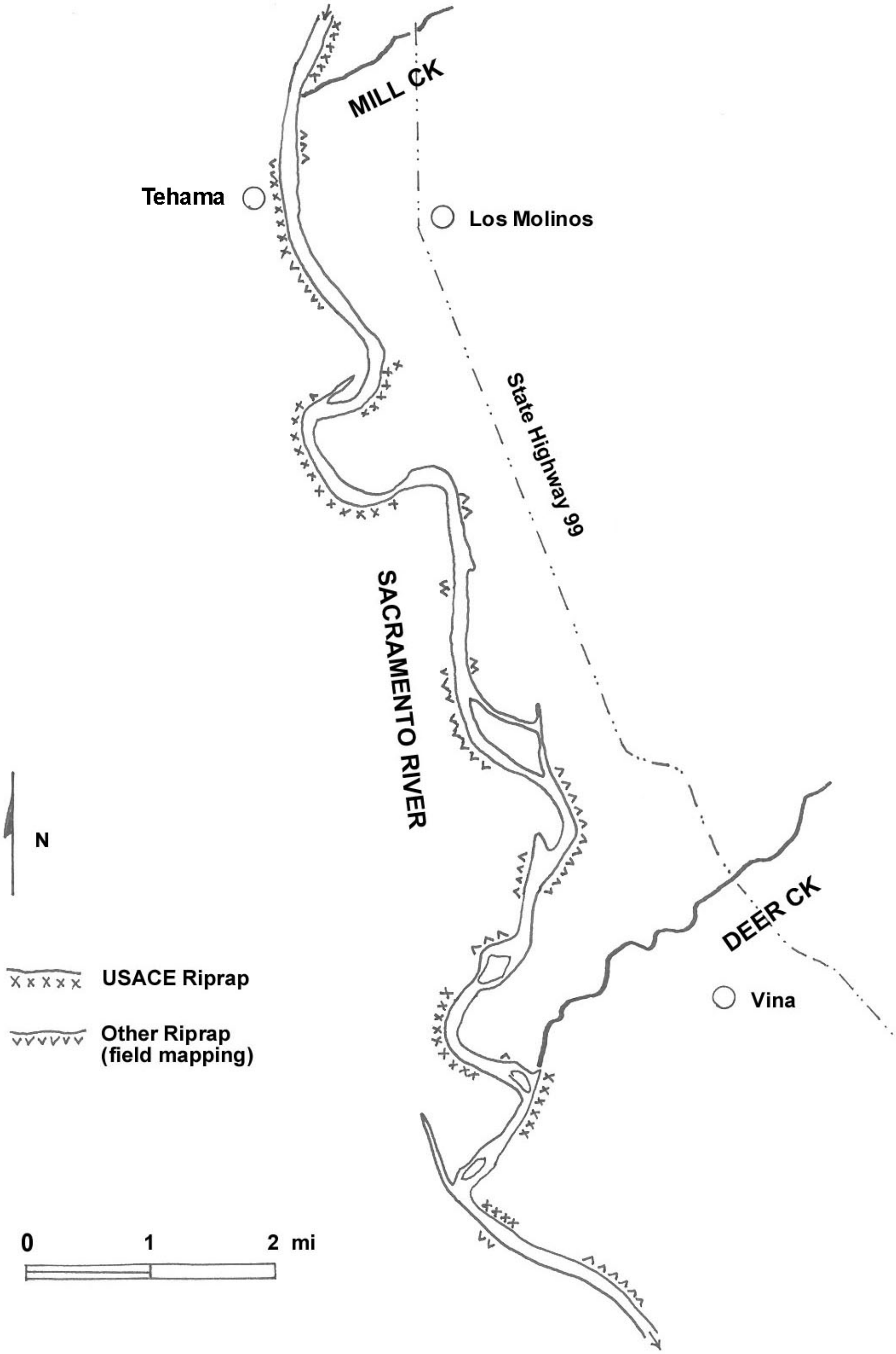


Figure 7-12. Map of riprap extent along the Sacramento River from the confluence of Mill Creek downstream to Woodson Bridge (RM 229-218) (adapted from unpublished map from Julie Cunningham, California Department of Water Resources, Red Bluff). Source: Figure 2.2-2 from Kondalf et al. 2000. Note that riprap is typically placed in locations most likely to provide bank swallow habitat (i.e., actively eroding meander bends), resulting in a disproportionately higher loss of swallow habitat (e.g., riprap on 50 % of bank length is likely to remove substantially more than 50 % of suitable bank swallow nesting habitat).

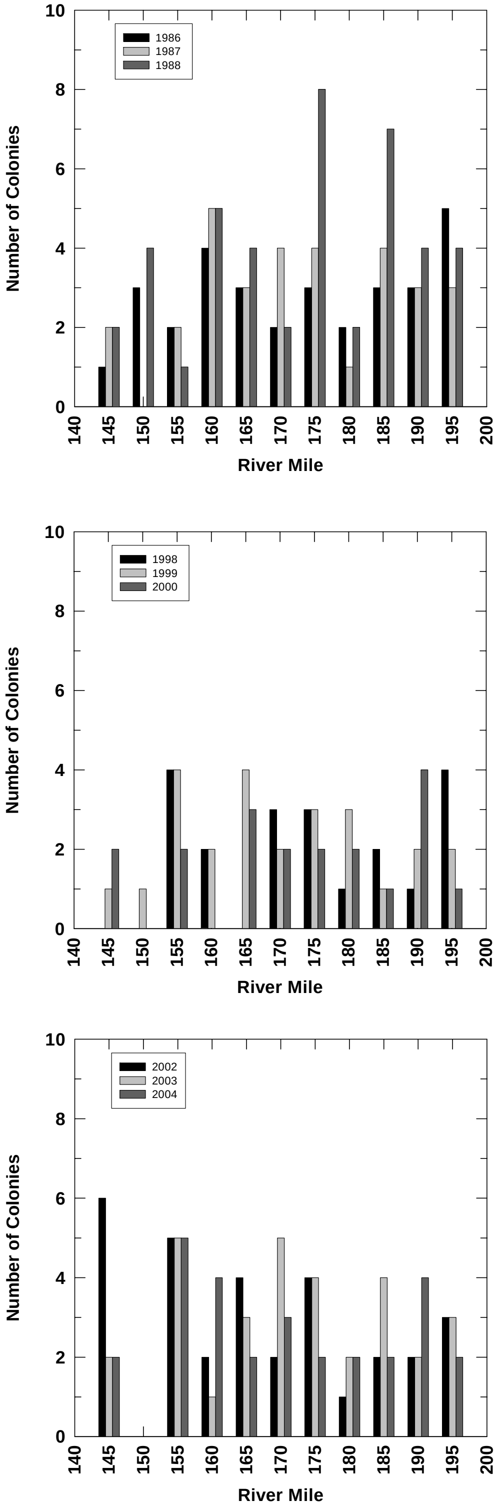


Figure 7-13. Longitudinal distribution of bank swallow colonies in the Sacramento River valley, grouped in 5 RM increments. Source: Garrison, 1989; Schlorff 1997, 1998, 1999, 2000, 2002, 2003, 2004.

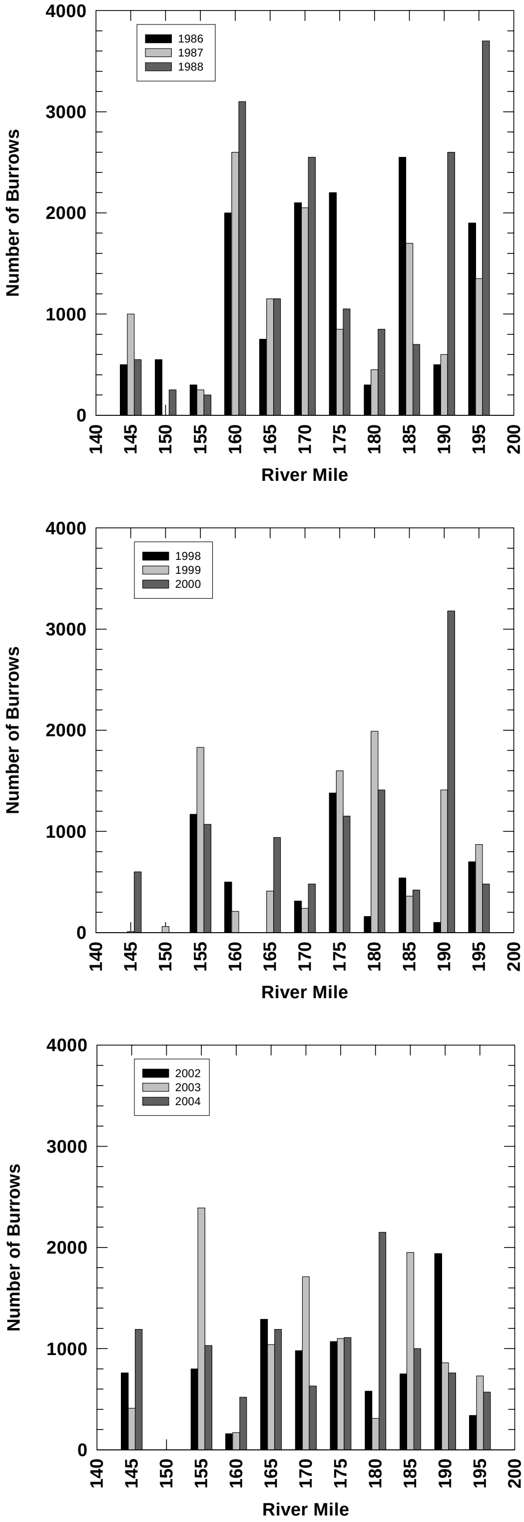
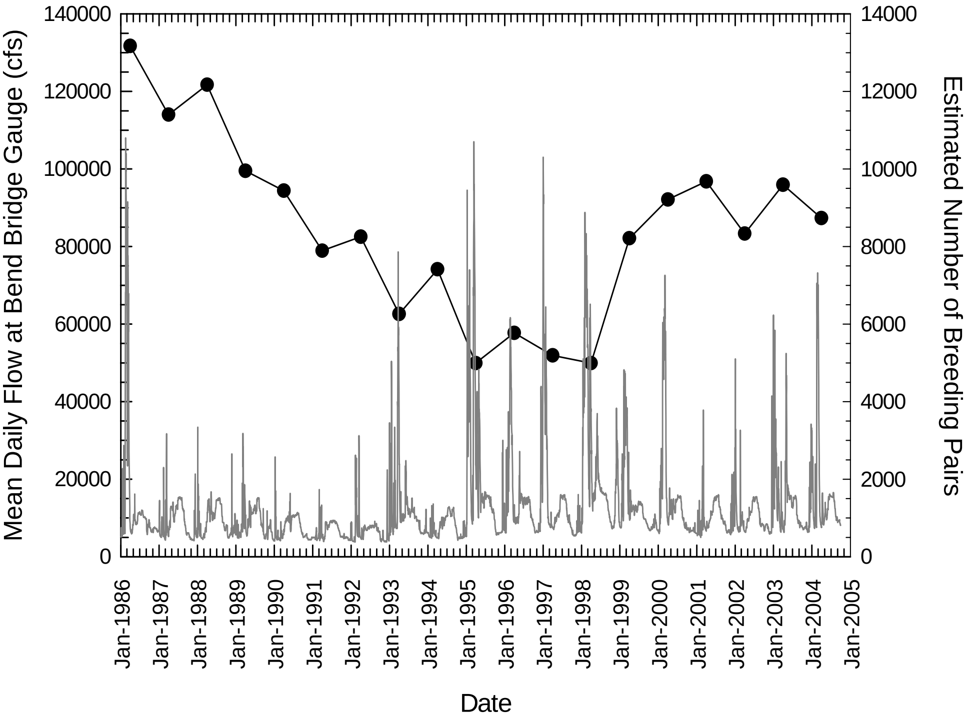


Figure 7-14. Longitudinal distribution of bank swallow burrows in the Sacramento River valley, grouped in 5 RM increments. Source: Garrison, 1988; Schlorff 1997, 1998, 1999, 2000, 2002, 2003, 2004.



Rm 200-243 (Hamilton City to Red Bluff)

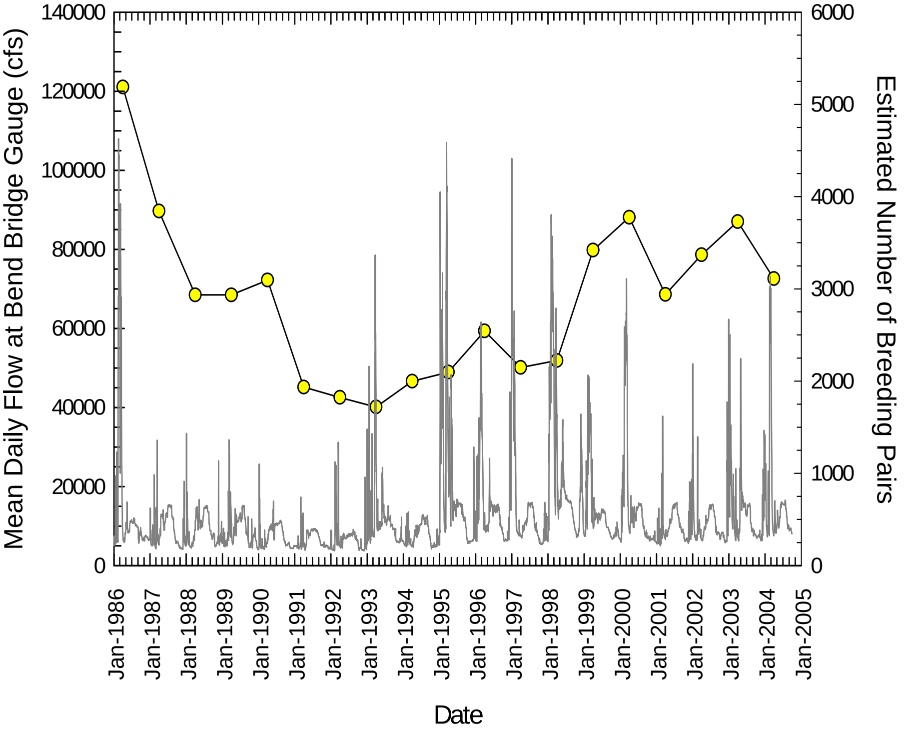


Figure 7-15. Mean daily flow at the Bend Bridge Gauge (gray line, left axis) at RM 258 and total estimated number of bank swallow breeding pairs (closed symbols, right axis) in (A) the Sacramento River as a whole and (B) Hamilton City to Red Bluff Reach (RM 200-243) during 1986-2004. Source: Hight 2000, Schlorff, 1997, 1998, 1999, 2000, 2001, 2002, 2003, 2004.

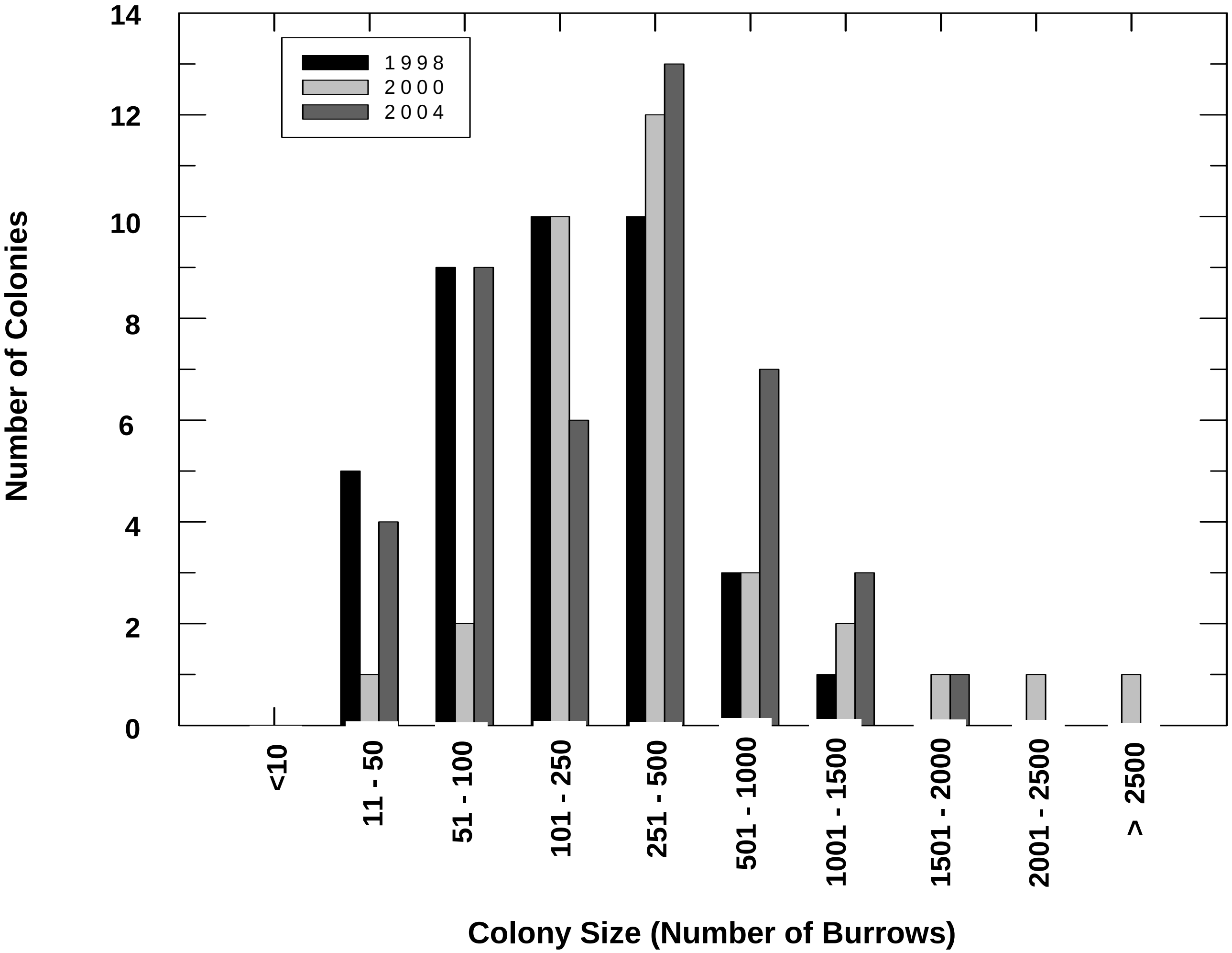


Figure 7-16. Frequency distribution of bank swallow colony sizes (total number of burrows per colony) in the Sacramento River valley for 3 recent survey years. Source: Schlorff 1998, 2001, 2004.

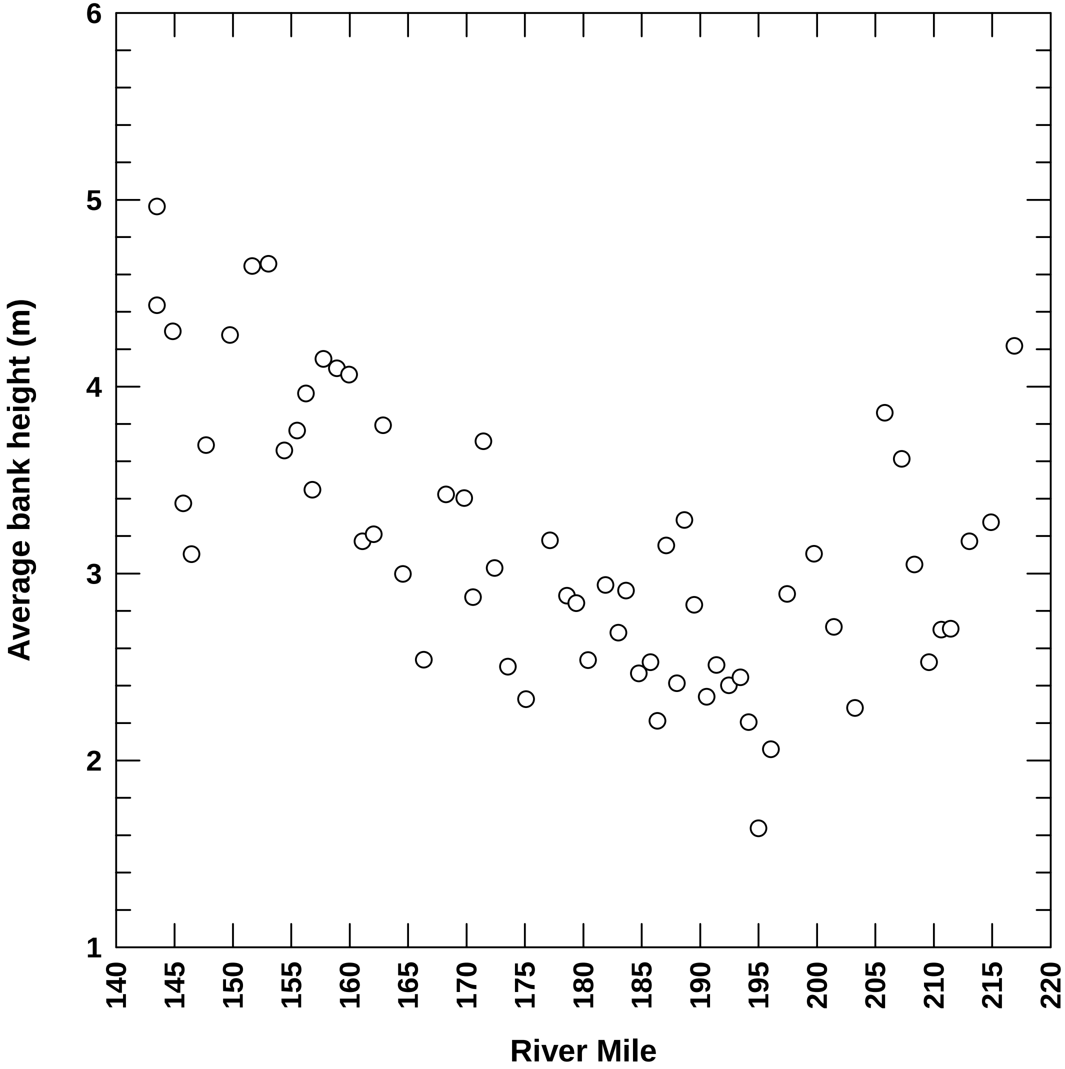


Figure 7-17. Mean bank height along the Sacramento River from Colusa to Woodson Bridge (RM 143 to 219). Data from Greco et al. *in review*.

8 WESTERN POND TURTLE

The western pond turtle (*Clemmys marmorata*) is California's only native freshwater turtle, and is considered a Species of Special Concern by the California Department of Fish and Game. The habitat needs of the western pond turtle are diverse. Along major alluvial river systems such as the Sacramento River, it uses oxbow lakes, sloughs, and other off-channel water bodies for foraging and rearing. Main channel habitats are used for aquatic dispersal and at least occasionally for foraging and basking. Western pond turtles also use upland areas, including grasslands, oak woodlands, and gaps in riparian forests, for nesting, dispersal, and overwintering. The aquatic and upland habitats need to be close enough that the turtles can readily access them both.

The relationships among physical processes, habitat conditions, and biological responses of western pond turtles are distinctly different from those of the other focal species considered in this SOS Report. Yet the habitats of western pond turtles are used by many species, which together contribute to the overall diversity of wildlife along the Sacramento River corridor. Maintaining this diversity will require maintaining the processes that provide a complex mosaic of off-channel and riparian habitats. Doing so will be predicated on an understanding not only of the processes themselves, but also of how they are affected by management actions. The western pond turtle was chosen as a focal species in the Sacramento River Ecological Flows Study (Chapter 1) because of it’s reliance on both off-channel aquatic habitats and adjacent upland habitats. As highlighted in the discussion below, the turtle's principal habitats are each affected by a unique set of geomorphic processes, such that the effects of land and water use on the western pond turtle (and other species that use the habitats) have been diverse and complicated in the Sacramento River system.

8.1 Geographic Distribution and Taxonomy

8.1.1 General distribution

The western pond turtle historically occurred in Washington, Oregon, and Baja California, and had a relatively continuous distribution within California principally west of the Sierra-Cascade crest (Buskirk 2002, Stebbins 2003). Western pond turtle populations are currently at a fraction of their historical levels (Holland 1994, Reese and Welsh 1997, Germano and Bury 2001, Stebbins 2003, and Bettelheim 2005). They nevertheless occur throughout much of their historical range (Stebbins 2003). Although a USFWS determination in 1992 found that western pond turtle listing under the ESA was not warranted (USFWS 1992), the species is listed as Endangered in Washington, Threatened in Oregon (Bettelheim 2005), and is considered a Species of Special Concern in California.

8.1.2 Local distribution

The Central Valley is thought to have supported the highest historical concentrations of western pond turtle (Holland and Bury, *in press*). An estimated 4 million turtles inhabited the region around Tulare Lake (Holland 1989, Bettleheim 2005), a 198,800 ha (486,400 ac) water body that has been almost completely replaced by dry farmland (Bettleheim 2005). The conversion of native wetlands and floodplains for urban and agricultural uses has eliminated most of the western pond turtle habitat of the Central Valley. This has contributed to widespread extirpation of the species in southern California (Germano and Bury 2001).

Western pond turtle numbers are greatly reduced, but the species is thought to still occur in rivers, backwaters, and wetlands of roughly 90% of its historical range (Jennings and Hayes 1994), including perennially flowing rivers in the Central Valley (Germano and Bury 2001, Holland 1994). The northern Sacramento Valley is estimated to support the species in abundance, but population data for the Central Valley in general and the Sacramento River in particular have rarely been collected. For example, the 2001 survey postdated the preceding survey by more than a decade. Moreover it did not consider potentially important habitats along mainstem rivers and within the adjacent floodplains (Germano and Bury 2001). Many of the western pond turtle populations that have been observed in recent years in the Central Valley appear to have recruitment levels and age structures that are consistent with maintenance of roughly stable populations (Germano and Bury 2001). However western pond turtles captured from the Sacramento River in 2003 by Dawn Wilson had a male-skewed sex ratio, relative to those captured on Big Chico Creek (Golet et. al. 2003). Male-skewed sex ratios may sometimes arise in turtle populations in areas of extensive surface road networks due to road kill of females associated with the gender-specific need for frequent travel to upland areas for nesting (Gibbs and Steen 2005). Expansion of agriculture and other development in upland areas has probably adversely affected nesting habitat and connectivity (Golet et. al. 2003, Gibbs and Steen 2005).

8.1.3 Taxonomy

Clemmys marmorata (also known as *Emys marmorata*, or *Actinemys marmorata*) has undergone numerous name changes since the species was first identified in 1852 (Bettelheim 2005, Buskirk 2002, Parham and Feldman 2002, Shaffer et al. 1997). Phylogenetic research has variously suggested that the western pond turtle may belong to any of three genera: *Emys*, *Actinemys*, and *Clemmys*. Recent molecular phylogenetics work points to the *Emys* genus, based on the evolution of shell kinesis in the western pond turtle (Buskirk 2002, Parham and Feldman 2002). Given that the *Clemmys* is not monophyletic, it is expected that some revision of the genus will probably occur after more extensive research is completed (Spinks et al. 2003).

In 1945, two subspecies of the western pond turtle were distinguished in California by M. Seeliger: the southwestern pond turtle (*C. m. pallida*) and the northwestern pond turtle (*C. m. marmorata*) (Buskirk 2002). This taxonomy is at odds with results from recent studies of molecular genetics, which suggest that western pond turtle populations fall into four distinct groupings or clades: 1) a Northern clade spanning the widest range, from San Luis Obispo County, California, to Washington, and including the northern Central Valley populations (Spinks 2005, Spinks et al. 2003, Buskirk 2002, Shaffer et al. 1997,); 2) a San Joaquin Valley clade in the southern Central Valley; 3) a geographically restricted clade in a short coastal stretch in Santa Barbara and Ventura counties, California; and 4) a Southern clade that ranges from the Tehachapi Mountains to Baja California, including areas to the west of the Transverse Ranges (Spinks 2005). Based on the results of the molecular genetics studies, the genetic variations in the southern California populations of western pond turtle have been described as cryptic, and are now the subject of ongoing research and consideration in conservation planning (Spinks 2005, Germano 2005).

8.2 Life History and Habitat Requirements

Although primarily an aquatic reptile, the western pond turtle needs terrestrial habitat for basking, overwintering, nesting, and traveling between ephemeral sources of water (Reese 1996). Available data do not provide any clear indication of what percentage overwinters in the mud (i.e., underwater) versus on land. In any case, its habitat requirements are diverse and are affected by the magnitude and frequency of flow on the Sacramento River in a complex variety of ways.

8.2.1 Life history

Breeding activity peaks in May through July (Table 8-1), but may occur throughout the year (Holland 1994, Reese 1996). Western pond turtles are philopatric, which implies that continuity of nesting habitat from year to year may be an important consideration. A tendency for clustering of nests has been noted and is poorly understood (Holland 1994). Western pond turtles have low fecundity, laying 1–14 eggs per clutch (Holland 1994, Reese 1996 Stebbins 2003). Two hatchling emergence patterns are shown in Table 8-1; the southern pattern reported for Central and Southern California populations and the northern pattern reported for populations in Northern California through Washington. It is currently unknown which pattern best applies to the Sacramento Basin population.

Table 8-1. Hypothesized timing of western pond turtle life stages along the Sacramento River.

Life stage	Month																							
	Jan		Feb		Mar		Apr		May		Jun		Jul		Aug		Sept		Oct		Nov		Dec	
Nesting									░	░	█	█	█	█	▒	▒	░	░						
Egg incubation									░	░	█	█	█	█	█	█	█	█	█	░	░	░		
Hatchling emergence – southern pattern															░	░	█	█	█	█	░	░		
Hatchling emergence – northern pattern			░	░	█	█	█	█	░	░														
Hatchling overwintering	█	█	█	█	▒	▒	░	░											░	░	█	█	█	█
Juvenile growth and adult activity					░	░	▒	▒	█	█	█	█	█	█	█	█	▒	▒	░	░				
Juvenile and adult overwintering	█	█	█	█	▒	▒	░	░									░	░	█	█	█	█	█	█

░	Period of low activity
▒	Period of moderate activity
█	Period of peak activity

The incubation period for western pond turtle eggs averages 80 days (mainly starting in June–July), but in some cases may exceed 100 days in California (Bettelheim 2005). Incubating eggs are extremely sensitive to increased soil moisture, which can cause high mortality (Bettelheim 2005, Shaffer 2005, Ashton et al. 1997). In wet conditions, (e.g., due to natural inundation, irrigation, and regulated high flows), eggs can literally explode from internal pressure caused by water absorption (Ashton et al. 1997). According to Lovich (1999), western pond turtles are subject to environmental sex determination, with males produced at lower incubation temperatures and females at higher temperatures. The pivotal temperature for the switch in sex appears to be approximately 86ºF (30ºC) (Ewert et al. 1994, as cited in Lovich 1999).

In colder climates, hatchlings may often overwinter in their nests, emerging in the following spring (Bettelheim 2005). In warmer climates, such as southern and central California, hatchlings tend to emerge from the nest in the early fall (Bettelheim 2005). Hatchlings spend much of their time in shallow water, within dense vegetation of submergent or short emergent macrophytes (D. Holland, pers. comm., as cited in Jennings and Hayes 1994). Hatchling and juvenile survivorship is considered to be low (Holland 1994).

Western pond turtles in California reach sexual maturity in 7 to 11 years. Survivorship for adults is thought to be high (Jennings et al. 1992). The western pond turtle has a potentially long lifespan; one recaptured individual is known to have survived at least 42 years in Trinity County (Jennings and Hayes 1994). Twenty-five years is generally considered to be the rough upper limit on age for most adults in natural settings (Bury 2005).

8.2.2 Habitat requirements

The western pond turtle inhabits a wide range of fresh or brackish water habitats including ponds, lakes, ditches, perennially filled pools of intermittent streams, and backwater and low-flow areas of perennial streams and rivers (Jennings and Hayes 1994). A key requirement is proximity to potential nesting sites. The relative importance to the regional western pond turtle population (or metapopulation) of flood basin and permanent pond habitats in upland areas versus the Sacramento River and floodplain off-channel habitats has not been established.

8.2.2.1 Nesting habitat

Although some general nesting habitat parameters have been quantified, data are sparse, particularly for the Sacramento River. Females build nests between 2.4 to 4.7 in (6 to 12 cm) deep, in dry clayey, loamy, or silty soils (Bettelheim 2005, Ashton et al. 1997, Reese 1996, Holland 1994, Rathbun et al. 1992), on gentle (< 15%), south- or west-facing slopes (Holland 1994), at distances ranging from 4.9 to 1,320 ft (1.5 to 402 m) (average=148 ft [45 m]) away from water (Holland and Bury *in press*, as cited in Spinks et al. 2003; Reese 1996; Nussbaum et al. 1983, Holland 1994). Nests are generally located in grassy meadows, away from trees and shrubs (Holland 1994), with canopy cover commonly less than about 10% (Reese 1996). There is no data available on the relative elevations of western pond turtle nests and water levels in adjacent water bodies for the Central Valley; these are important habitat parameters that require further research (B. Bury, pers. comm., 2005; D. Germano, pers. comm., 2005).

8.2.2.2 Aquatic habitat

Western pond turtles are not especially strong swimmers. Suitable aquatic habitats generally have standing (lentic) and slow-moving (lotic) water, which, on the Sacramento River and other large, lowland alluvial rivers typically occurs in off-channel areas, such as oxbows and sloughs (D. Germano, pers. comm., 2005). This has been confirmed locally by anecdotal evidence, and by surveys in three oxbow lakes in the middle Sacramento River (Golet et al. 2003). For example, it has been noted that western pond turtle populations on the Sacramento River near Chico are concentrated away from the mainstem, in sloughs and oxbows, where they are generally less affected by the river's flow fluctuations (J. Campbell, pers. comm., 2005). A lack of association of western pond turtles with levees and rip-rapped areas has also been noted in the Chico area (J. Campbell, pers. comm., 2005).

Oxbows that are better connected to the hydraulics of the active river are also more strongly affected by fluctuations in mainstem flow. However, this kind of flow variability is probably not an important regulator of oxbow habitat quality for western pond turtles, because juveniles and adults can readily abandon areas that become unsuitable and search for better aquatic habitats (D. Germano, pers. comm., 2005) although overland travel may expose them to increased risk of predation (Gibbs and Steen 2005, Ashton et. al. 1997, Reese 1996). Overwintering in terrestrial habitats may be an adaptation which helps western pond turtles escape high winter flows in lotic waters (Ashton et al. 1997). It seems likely that even adults could be adversely affected by exceptionally high winter flows. However the extent to which the various lifestages of the western pond turtle are affected by high winter flows is currently unknown but is nevertheless an important consideration.

Western pond turtle, being relatively weak swimmers, can be easily displaced downstream by fast moving water (Ashton et al. 1997). The mainstem channels of large rivers are therefore not generally expected to provide optimal habitat for western pond turtles (Reese and Welsh 1998b; D. Germano, pers. comm., 2001; B. Bury, pers. comm., 2005). Water velocities of the mainstem Sacramento River in particular are probably too high for western pond turtles (D. Germano, pers. comm., 2005). Observations from the Trinity River and other large rivers in the Pacific Northwest appear to support the hypothesis that there is an inverse relationship between river size (measured by stream order) and density of western pond turtles in mainstem habitats (Reese and Welsh 1998b).

Western pond turtles that do occur in larger river systems are generally concentrated in side-channels and backwaters, and typically migrate to off-channel habitats, such as oxbows, during high flows (Holland 1994, Ashton et al. 1997). These western pond turtles may also overwinter, generally for 1–2 months, but sometimes for up to 6.5 months, in upland areas under leaf litter (Reese and Welsh 1997, Buskirk 2002, Bettelheim 2005). On the Trinity River, in un-dammed riverine habitat, western pond turtles appear to prefer deep, lotic water, moderate amounts of riparian vegetation, warm water and/or ample basking sites, and LWD and rocks (Reese 1996, Reese and Welsh 1997, 1998a, 1998b) which provide underwater cover from predators such as otters and minks. The limited use of mainstem habitat on the Sacramento River by the western pond turtle is a key factor limiting overlap between its primary habitat and those of the other focal species in considered in the SOS report.

Canopy cover in both riverine and off-channel habitats is thought to provide western pond turtles with protection from avian predators. Documented canopy associations for juveniles and adults include immature riparian vegetation (i.e., early-seral stage willow scrub) and canopy cover levels ranging from those of unvegetated gravel bars to those of mature, late-seral stage riparian vegetation (Reese 1996, Reese and Welsh 1998b).

In addition to physical habitat conditions, predation pressure has been shown to influence the distribution of western pond turtles. A case in point comes from studies in the San Simeon area of coastal California, in which fewer western pond turtles were observed when raccoon numbers were high. Raccoons are an important predator of western pond turtles and are known to prey on adults as well as juveniles (D. Germano, pers. comm., 2005).

8.2.2.3 Habitat for hatchlings and juveniles

Whereas adults and older juveniles are considered aquatic habitat generalists, hatchlings and young juveniles require specialized habitat for survival through their first few years. For example, in addition to requiring low-flow and backwater areas of rivers, hatchlings need to spend much of their time feeding in shallow water amongst dense submergent and short emergent vegetation, presumably to avoid predators (D. Holland, pers. comm., as cited in Jennings and Hayes 1994). Young western pond turtle growth rates are thought to be closely tied to food abundance, particularly the concentration of zooplankton fauna in the water column (Jennings and Hayes 1994, Holland 1994). Habitats preferred by juveniles are generally scarce and may be especially sensitive to anthropogenic and natural disturbances (Jennings et al. 1992). The extent to which this is the case on the Sacramento River is unknown due to a lack of observational data.

8.2.2.4 Basking habitat

Western pond turtles are poikilothermic ("cold-blooded") and generally must spend a portion of each day basking (Jennings and Hayes 1994, Zeiner et al. 1988), either on land or in thermal aquatic refugia. Terrestrial basking sites may include rocks, logs, banks, emergent vegetation, root masses, open banks, and tree limbs (Reese 1996, Reese and Welsh 1998b, Zeiner et al. 1988). Deep (> 1.6 ft [0.5 m]), still water with emergent woody debris, overhanging vegetation, and rock outcrops provide optimal basking habitat for older western pond turtle life stages (Bury 1972). Terrestrial basking promotes synthesis of vitamin D, controls parasites (Reese 1996), and permits the turtles to thermoregulate, thus assisting digestive processes.

Basking in water can permit western pond turtles to attain body temperatures of up to 86–93ºF (30–34ºC) for several hours per day, even in relatively cold streams (Bury 2005). A comparative study of aquatic basking in the Trinity River system showed that turtles in colder waters seem to spend more time seeking aquatic thermal refugia and basking than turtles in warmer waters (Bettaso 2005).

Observations from the Russian River (Sonoma County, California) indicate that typical western pond turtle basking sites have water depths of 3.2–6.6 ft (1–2 m) (average=4.76 ft [1.45 m]) and include some overhead riparian canopy cover, with more than 64% of observed sites having canopy cover of 20% or greater (Cook and Martini-Lamb 2005). Basking western pond turtles on the Russian River seem to prefer live, downed trees or new snags over older, worn or decomposed snags (Cook and Martini-Lamb 2005). Juveniles and adults of both sexes (Bettelheim 2005) have been known to compete aggressively for basking sites (Nussbaum et al. 1983) by biting, pushing, and making open-mouthed threats (Bettelheim 2005).

Warm summer air temperatures in the Central Valley may make thermoregulation activities of western pond turtles less important than they are in colder environments (Germano and Bury 2001). Moreover, for much of the year, western pond turtles in the Central Valley may be able to reach suitable body temperatures by basking in beds of aquatic vegetation (e.g., algae or vascular aquatic macrophytes). Dense beds of aquatic macrophytes can create thermal stratification, with warmer water in vegetated areas near the surface (Collins et al. 1985). By remaining within warm stratification layers western pond turtles can meet thermoregulation requirements while remaining in cover that reduces risk of predation. Germano and Bury (2001) observed a significant number of turtles in Dry Creek using algal mats for thermoregulation.

8.2.2.5 Temperature requirements

Temperatures preferences and requirements of western pond turtles are not well understood. Adults do not seem to allow body temperatures to exceed 93ºF (34ºC) (Lovich 1999), and also seem to avoid water temperatures greater than 102–104ºF (39–40ºC) (D. Holland, pers. comm., as cited in Jennings and Hayes 1994). Data from the Trinity River indicates that juveniles tend to occur in a wide range of water temperatures between 54–91ºF (12–33°C), whereas adults appear prefer a narrower range of water temperatures between 50–63ºF (10–17°C) (Ashton et al. 1997). Water temperature appears to have a strong effect on activity levels of western pond turtles, with notably higher activity in water temperatures that consistently exceed 59ºF (15ºC) (Jennings and Hayes 1994). Downstream of dams with hypolimnetic summer flow releases, temperatures are generally much cooler than they would be under natural conditions, and may result in unnaturally slow western pond turtle growth rates, thus affecting body size and age at maturity (Reese and Welsh 1997, 1998a). A case in point comes from studies on the Trinity River, where water temperatures in the regulated mainstem are more than 50ºF (10ºC) colder than they are on the unregulated South Fork Trinity River (Ashton 2005), and where the expression of western pond turtle phenotypic traits in colder reaches appears to be such that stunted turtles reach sexual maturity at an unnaturally early age (Ashton 2005). Data from a separate series of studies on the Trinity River suggest that, in regions with cold winters, western pond turtles generally take refuge from the main river from October or November until April or even later (Reese 1996, Reese and Welsh 1998a), with a majority seeking terrestrial overwintering sites and a smaller fraction choosing lentic aquatic sites (Reese and Welsh 1998a, Reese 1996, Holland 1994).

8.2.3 Synthesis of life history and habitat requirements

The western pond turtle is a freshwater habitat generalist that requires terrestrial habitat for nesting. It occurs in a wide range of standing (lentic) and low-velocity moving (lotic) waters, in freshwater habitats that are close to suitable terrestrial nesting habitat. Typical freshwater habitats include ponds, lakes, streams, rivers, side channels, oxbow lakes, wetlands, wastewater treatment ponds, and canals. The western pond turtle is a poor swimmer and generally avoids the fast moving waters found in the main channel of large rivers such as the Sacramento.

The western pond turtle is a poikilothermic (“cold-blooded”) reptile that basks to regulate body temperature. Terrestrial basking typically occurs on logs, banks, and rocks, whereas aquatic basking occurs in areas with warm tributary or groundwater inputs, in backwater areas that are shallow enough to be warmed by solar isolation, and in aquatic macrophyte beds where thermal stratification keeps the upper part of the water column warm in spring and summer. Juvenile and adult western pond turtles are most active during the warmer months of the year (i.e., from spring through fall), when individuals are generally more able to keep body temperatures high enough for foraging, escaping predation, mating, oviposition, and dispersal.

Metabolic demands are positively correlated with body temperature (as well as food supply) up to a critical temperature (i.e., the threshold at which serious lethal or sub-lethal temperature effects appear). For juveniles, growth rates should generally increase with temperature, as long as food supply is abundant. Faster juvenile growth rates should provide a selective advantage by reducing the window of vulnerability of small juveniles to size-limited predators. Because turtles can readily move among water bodies, particularly at the local scale, it is expected that selection should favor individuals that actively choose warmer, food-rich aquatic sites that are located near suitable terrestrial nesting sites.

Optimal conditions for western pond turtles should occur in permanent lentic habitats that support a mosaic of dense submergent or emergent macrophytes, with some open water zones in deeper areas (for cover, refuge, and persistence of water during droughts), abundant zooplankton, other aquatic invertebrates, and vegetation (all for food), appropriate cover (with deep water and dense vegetation) and aquatic basking sites (in thermal stratification zones associated with macrophyte beds). Proximity to riparian forests gives sites such as these the added advantage of inputs of large wood that can provide additional aquatic cover (when wood falls into water) and basking sites (when wood falls near shorelines). Oxbow lakes and abandoned channels or sloughs on the Sacramento River sometimes have all of the above conditions.

See Table 8-1 for a summary of the expected life history timing and Table 8-2 for an overview of the key factors (required habitat elements, food, predators, other biotic factors) affecting each life stage for western pond turtle populations in the Sacramento Valley.

Table 8-2. Summary of key factors (habitat elements, food items, predation, and other biotic interactions) expected to affect each life history stage of western pond turtle populations in the Sacramento Valley.

Life history stage	**Key factor (NA = not applicable)**							
	Terrestrial (upland)	**Aquatic (lentic or backwater)**	**Food supply**	**Aquatic cover**	**Aquatic basking**	**Terrestrial basking**	**Predators**	**Other biotic interactions**
Nesting	• Soil = clay, silt, loam; gentle slope • Elevated high (and dry) above nearby water during incubation • Temperature = *f(ambient temp. & microclimate, aspect, topographic and vegetative shading)* • rarely in canopied areas	• NA	• NA	• NA	• NA	• NA	• Raccoons • Coyotes	• NA?
Incubation	• Low soil moisture from June to October (lack of inundation or high precipitation during incubation) • Incubation lasts 80–100+ days	• NA	• NA	• NA	• NA	• NA	• Raccoons • Coyotes	• NA?
Hatchling (neonate)	• NA if emerge early from nest (?) • Protection from high flows and predators if overwintering in nest • Leaf/duff suitable for aestivation (if remaining in nest)	• Low-velocity, warm water areas, with at least some aquatic vegetation for cover • Shallow edgewater/ backwater areas with little or no flow	• Zooplankton • Small aquatic invertebrates • Aquatic vegetation (?)	• Dense submergent and short emergent vegetation in shallow water	• Emergent vegetation areas, shallow edgewater • Will use small woody debris or rocks near shore in shallows, require shallower	• NA (?)	• Raccoons • Bullfrogs • Fish • Birds	• Competition with fish for zooplankton or other invertebrates?

Life history stage	Key factor (NA = not applicable)							
	Terrestrial (upland)	**Aquatic (lentic or backwater)**	**Food supply**	**Aquatic cover**	**Aquatic basking**	**Terrestrial basking**	**Predators**	**Other biotic interactions**
	or cover (if early emergence occurs) • Proximity to suitable feeding/aquatic habitat for shelter				warmer water than juveniles or adults			
Juvenile	• Overwintering (?) • Basking areas • Some canopy cover (immature riparian vegetation–late seral stage riparian vegetation)	• Warm water and low water velocity areas, ponds adjacent to riverine habitat • Side channels and oxbows	• Aquatic invertebrates • Small fish • Carrion • Aquatic vegetation	• Algal beds and other aquatic macrophytes (vascular plants)	• Basking areas: emergent vegetation, rocks, LWD, shallow edgewater areas with low bank gradients (prefer woody perches over rocky or vegetated perches) • Prefer aquatic basking sites over terrestrial basking sites (more protection from predation and quicker escape) • Lower flow (less water velocity) areas than adults	• Logs and other woody debris • Sheltered banks with low gradients • Boulder/bedrock or other mid-channel island areas	• Raccoons • Birds • Bullfrogs?	• Competition with fish for aquatic invertebrates and other food?
Adult	• Overwintering: thick duff/leaf litter in upland habitats up to 500m from watercourse • Dispersal (distances over 1	• Side channels and oxbows, lower flow areas • Overwintering in mud at the bottom of aquatic ponds • Active season (breeding) habitat	• Aquatic invertebrates • Amphibian larvae • Fish • Carrion • Aquatic	• Algal beds and other aquatic macrophytes (vascular plants)	• Basking areas: emergent vegetation, rocks, LWD, shallow edgewater areas with low bank gradients (prefer	• Logs and other woody debris • Sheltered banks with low gradients • Boulder/bedrock or other mid-channel island	• Raccoons	• Competition with fish aquatic invertebrates and other food? • Competition w/ introduced

Life history stage	Key factor (NA = not applicable)							
	Terrestrial (upland)	**Aquatic (lentic or backwater)**	**Food supply**	**Aquatic cover**	**Aquatic basking**	**Terrestrial basking**	**Predators**	**Other biotic interactions**
	km) • A seasonal terrestrial habitat use during non-winter periods • Basking areas • Some canopy cover (immature riparian vegetation–late seral stage riparian vegetation)	areas: basking areas, refugia, and foraging habitat • Refugia: Undercut banks, LWD, submerged vegetation, rocks • Foraging: oxbows, shallow, edgewater habitats, aquatic macrophyte beds (e.g., algae)	vegetation		woody perches over rocky or vegetated perches) • Prefer aquatic basking sites over terrestrial basking sites (more protection from predation and quicker escape)	areas		turtles for food and basking sites? • Disease • Parasites?

8.3 Conceptual Model of Historical Conditions in the Sacramento Valley

Under historical conditions in the Sacramento Basin, primary habitat for western pond turtles should have occurred in two general zones:

oxbow lakes and abandoned channels or sloughs within the broad riparian zone that extended from RM 245 (near the current location of Red Bluff) to the Delta, and

the seven large flood basins containing complexes of tule-dominated wetlands and distributary channels that bordered the outer margins of the riparian zone on both sides of the Sacramento River from about RM 180 (just south of Ord Bend) downstream to the Delta (see Figure 8-1).

Many of the wetlands would have dried by late summer in a normal or dry year. However, deep off-channel habitats (i.e., oxbow lakes and sloughs) in the floodplain and distributary channels in flood basin wetlands would have retained water throughout the summer, thus providing suitable habitat for juvenile and adult foraging and cover. Accurate estimates of suitable aquatic habitat area for western pond turtles are not available, but the historical wetlands area for the Sacramento Basin has been roughly estimated at approximately 122,000 ha (301,000 ac), with about 35,000 ha (87,000 ac) in riparian zones and 87,000 ha (214,000 ac) in flood basins (The Bay Institute 1998).

The main channel of the Sacramento River would have provided secondary habitat along low velocity margins, in backwaters, and within the shallow, braided side channels, which were historically more abundant in the first quarter of the 20th century. High velocities would generally have excluded turtles from large portions of the river, but the main channel could have provided relatively more habitat when flows dropped and waters warmed during the summer low flow period. This may have aided dispersal and gene flow among Sacramento River off-channel habitats and tributaries as well. Nesting habitat would have occurred in native grasslands or oak woodlands found on higher terraces and on natural levees in the riparian zone, in the upland grasslands and woodlands along the edges of the flood basins, and possibly in smaller herbaceous openings within the riparian forest. It is possible that western pond turtles also nested in the upper sandy ends of point bars that happened to be close to off-channel water bodies, as they have been observed to do in the more arid Mojave River ecosystem. Egg mortality from flooding would have been very unlikely in most of these habitats given the steady low base flows of the river during summer and early fall: mean monthly flows in June and July typically averaged 4,000 to 7,000 cfs, while flows exceeding 15,000 cfs during the summer (June–September) were rare (Figure 2-4, Kondolf et al. 2000, CDWR 2001). A flow as high as even 15,000 cfs would be unlikely to inundate nests, because it would be well within the active channel and thus below likely nesting sites (unless some turtles nested on sandy point bars rather than on higher floodplain surfaces). If a rare, high flow event did occur during a wet spring-summer, the local populations could have readily withstood the loss of an individual egg class. Given the long lifespan of the western pond turtle, population viability or sustainability would not have been noticeably affected by such year class losses long so they occurred, on average, in only 1 year (or fewer) out of 10 (D. Germano, pers. comm., 2005).

Western pond turtles would have been expected to occur upstream of RM 245, but the more confined river and greatly reduced channel meander migration zone that characterizes this section of the river would have limited off-channel habitats. Summer water temperatures in the vicinity of RM 300 were likely in the 59 to 68ºF (15 to 20ºC) range, and relatively low summer base flows would have created conditions suitable to support western pond turtles—but probably at much lower densities than in floodplain and flood basin habitats in the reach below RM 245.

The metapopulation of western pond turtles within the Sacramento Valley would have been well-distributed historically, with larger source populations in the riparian and flood basin zones described above, and smaller or lower density populations along all or most major tributaries and the Sacramento River upstream of RM 245. Under these conditions the western pond turtle metapopulation should have been able to withstand periodic disturbances, including complete loss of smaller satellite populations and infrequent bouts of high mortality of eggs, hatchlings, juveniles, and possibly adults that might have occurred during years with exceptionally high flows.

8.4 Conceptual Model of Western Pond Turtle Habitat Dynamics

To provide the context necessary for understanding how anthropogenic changes have affected western pond turtle habitat (and to develop management actions designed to compensate for any adverse effects), it is necessary to develop a conceptual model for how key habitat elements would have evolved under natural conditions. Figure 8-2 is a graphical summary of such a conceptual model, showing the cyclical evolution of a main channel meander bend to oxbow slough and lake, then to riparian forest, and eventually back to a new main channel. This model seeks to convey how patterns of flow, sediment transport, vegetation succession, and geologic constraints interact to regulate meander migration and sediment deposition, which together are responsible for the creation and evolution of oxbow sloughs and lakes—the prime habitat for western pond turtles on the Sacramento River. Chapter 3 includes much of the supporting background material that was considered in development of Figure 8-1.

As shown in Figure 8-1, the main channel meander bend is eroded by flows that overcome the bank's resistance to erosion, where that resistance is determined by a complex interaction among many factors including geologic constraints, material properties of the bank, and riparian root systems. Progressive migration occurs, causing an increase in sinuosity until meander geometry is such that the river would, under certain high flow conditions, be able to excavate a cutoff channel through the bend, and thus create a new main channel.

At some point the high flow threshold is exceeded and a chute cutoff occurs, with the river occupying a new channel and a flowing side channel occupying the former main channel. The side channel is eventually converted into an oxbow slough when the upstream end is disconnected from the mainstem by deposition of a gravel plug, generally in the receding stages of a high peak flow. The oxbow slough then eventually becomes a lake when a gravel plug is deposited on its downstream end.

Inundation of the oxbow lake by periodic high flows leads to sediment deposition, which, together with influxes of organic debris from both newly established vegetation and more mature vegetation on surrounding higher elevation surfaces, eventually fills the lake, turning it into an emergent wetland (dominated by tules [*Scirpus* species or other herbaceous emergent wetland plants]) or swamp (dominated by buttonwillow [*Cephalanthus occidentalis*] or willows [*Salix* spp.]). With continued sedimentation, organic infilling, and vegetation succession, the wetland becomes a riparian forest. Throughout the evolutionary process in the abandoned channel, the mainstem of the river progressively migrates elsewhere, subject to geologic constraints, and eventually may work its way back to its old course, resetting the process by eroding the infilled oxbow lake.

Lenses of clays or silts from past infilling of oxbow lakes creates local patterns of soil texture and stratification within a more dominant matrix of alluvial sands and gravels. This creates a heterogeneous physical template in the river's floodplain and affects rates of physical processes such as channel migration (see Chapter 3), patterns of vegetation recruitment and successional dynamics (Chapter 9), and habitat suitability for western pond turtles and other species such as bank swallows (Chapter 7).

8.5 Effects of Changes in Western Pond Turtle Habitat

This section identifies the primary changes from historical condition and the associated anthropogenic land and water uses that are hypothesized to have had a significant effect on the current conditions for western pond turtles in the study area. A conceptual model of current conditions is presented in Section 8.6 to summarize and synthesize the effects of these changes, and provide guidance to development of appropriate management strategies to maintain and enhance western pond turtle populations and their habitats in the Sacramento Valley.

8.5.1 Reclamation of flood basin and floodplain areas

Reclamation of flood basin and floodplain areas for agricultural uses has led to a poorly quantified but potentially sharp decline in western pond turtle populations and their habitats along the Sacramento River and its historic floodplain and flood basins.

Early reclamation activities began in the late 1800s, focusing primarily on natural levees and riparian-floodplain areas, and included in-filling of oxbow lakes and sloughs and the removal of native vegetation. These efforts would have generally affected smaller (easier to fill) off-channel habitat units, resulting in isolated losses in wetland, lentic, and other off-channel aquatic habitats within the riparian zone. In many cases, however, reclamation was more extensive, with leveling activities that affected many acres of floodplain, resulting in complete loss of aquatic habitat value, and alteration of the surrounding riparian forest (The Bay Institute 1998). The extensive marshes and poorly drained soils of the flood basins delayed serious land reclamation efforts until around 1910, when construction of the Sacramento Flood Control Project helped industrialize rice farming in the Sacramento Valley leading to widespread loss of natural flood basin wetlands (The Bay Institute 1998).

The net effect of reclamation activities on historical aquatic habitats has not been fully quantified for the seven flood basins of the Sacramento River, but it is known to be severe, amounting to a nearly complete loss of large flood basin wetlands and lentic habitats (The Bay Institute 1998). The best available information suggests that flood basin and riparian wetlands have been reduced by about 90% (The Bay Institute 1998). Limited amounts of potentially suitable western pond turtle aquatic habitat do, however, remain in some agricultural canals and managed wetlands in the National Wildlife Refuges (Lechner and Wilson 2004, Germano and Bury 2001). The potential value of inundated habitats in the Yolo or Sutter Bypasses for western pond turtles is currently unknown. Eight perennial ponds (including both natural and engineered ponds) currently occur in the Yolo Bypass and support assemblages of non-native and native fishes (Feyrer et al. 2004), but their potential to support western pond turtles has not been studied. Losses of potential terrestrial nesting habitat due to reclamation have probably been significant as well, both within the historical flood basins and the riparian zone. Riparian forests are estimated to be at only 5–10% of historical levels (Katibah 1984, The Bay Institute 1998, Greco 1999, see also discussion in Chapter 9). Most of the lost acreage of forests has been converted to agriculture (row crops or orchards) or other human development that would, at best, be expected to provide reduced quality nesting habitat.

In addition to direct loss of habitat, there have undoubtedly been substantial, but unquantified and relatively unstudied, impacts to the quality of current aquatic and terrestrial western pond turtle habitat caused by general habitat fragmentation, partial or complete movement barriers (e.g., roads, levees, riprap, steep bank slopes along many canals), and other anthropogenic disturbances, such as flood irrigation or plowing, and possibly other management activities such as herbicide and pesticide spraying, within orchards or agricultural fields that might otherwise be suitable for nesting.

8.5.2 Changes in the frequency and magnitude of floods

We hypothesize that changes in the frequency and magnitude of floods have had substantial direct and indirect effects on the quality and quantity of western pond turtle habitats on the Sacramento River. Flow regulation of the Sacramento River has increased summer base flows (to satisfy downstream irrigation demands) while reducing the magnitude of winter flood peaks (to satisfy to dam storage requirements and to mitigate downstream effects of flood flows) (Figure 8-3) (CDWR 1980, Buer 1984, Lowney and Greco 2003). The reduction in flood peaks emerges clearly from a comparison of annual peak flows. For example, the mean annual flood has been reduced from approximately 110,000 cfs to 74,000 cfs based on analysis of annual peak flows from the period 1903–1943 and 1944–2002 (Larsen et al., unpublished).

By reducing the magnitude and frequency of peak winter flows, regulation at Shasta and Keswick Dams has also reduced the magnitude and duration of overbank flows. This in turn has reduced the potential energy available for scour and deposition of sediment and has thus restricted the river's ability to generate and maintain off-channel habitats important to western pond turtles. However, the relationship between cutoff frequency and overbank flow appears to have changed in the post-dam interval such that higher cutoff frequencies are produced at a given cumulative overbank flow (Figure 3-12). As discussed in Chapter 3, this may be due to the effects of agricultural clearing on floodplain surfaces. The implication for western pond turtle habitat is that cutoff frequency has increased even as overbank flow has been reduced. However, as shown in Table 3-9 and Figure 3-10, mean sinuosity of cutoff bends has decreased over time as the frequency of cutoff has increased.

Reductions in overbank flow have important implications for succession of riparian vegetation, and thus have indirect effects on western pond turtles, which depend throughout much of their life history on varying levels of canopy cover. Moreover, riparian vegetation helps regulate the roughness and cohesiveness of floodplain surfaces, and thus helps regulate hydraulics in a way that may fundamentally influence where and how secondary channels are formed (Micheli et al 2004). If vegetation succession in native riparian forest is interrupted by flow regulation, it might have important implications for off-channel habitat dynamics (Greco and Plant 2003).

Relationships between overbank flow and secondary channel formation (via scour) or filling (via deposition) remain poorly understood. While reduced energy available for scour would be predicted to decrease the availability of off-channel habitat in the long term, reduced incidence of overbank flow may serve to extend the lifespan of existing off-channel habitats by reducing sediment deposition rates. A reduction in overbank flow may nevertheless be reasonably predicted to reduce the overall dynamism, complexity and diversity of off-channel habitats.

8.5.3 Changes in sediment supply

Reductions in sediment transport rates, attributable to retention of sediment behind Shasta and Keswick dams (CDWR 1980, Singer and Dunne 2001, CDWR 1994), have changed sedimentation rates in existing off-channel water bodies, and have thus altered the lifespan of available western pond turtle aquatic habitats required for foraging and cover during the spring and summer activity period. Deposition of fine sediment in particular drives the process of terrestrialization of oxbow lakes and sloughs (Piegay et al. 1998, Morken and Kondolf 2003). Reduced frequencies of overbank flow, combined with reduced suspended sediment concentrations, may serve to extend the life span of off-channel habitats. The extent to which this is offset by any increase in sediment loading from agricultural runoff is difficult to determine in the absence of data. Additional quantitative data about sediment loading to off-channel habitats would help shed light on this issue.

Decreased sediment loading may have had particularly significant effects on oxbow lakes, which are disconnected from the mainstem and thus may only aggrade (fill-in) during the largest, most infrequent overbank flow events. Dam-related reductions in suspended sediment delivery to the floodplain have probably also reduced the river's capacity to deposit material on the floodplain and thus create topographic variation and complexity. This would tend over the long-term to result in an unnaturally static distribution of oxbow lakes and sloughs.

This pattern of static off-channel habitats would tend to be reinforced by dam-related reductions in bedload transport, if it has locally reduced bar aggradation rates and thus locally decreased meander migration rates. Recall from Chapter 3 that bar aggradation (which will increase if excess bedload becomes stored in bars) is thought to be a localized accelerator of progressive migration rates on the Sacramento River (Constantine et al., unpublished). Hence, under regulated conditions, with reduced bedload transport and reduced aggradation, progressive migration rates may locally be lower than under unregulated conditions.

In addition to locally affecting meander migration rates, gravel bar dynamics can also regulate the connectivity of off-channel habitat to the mainstem, and thus alter its quality for western pond turtles. For example, the primary mechanism of closure of abandoned meanders from the main channel (a necessary step in the formation of oxbow lakes) is the deposition of a gravel plugs that are derived from passing bedload (Teisseyre 1977, Piegay et al. 1998, Morken and Kondolf 2003). Hence, reduced bedload under regulated conditions may be less likely to generate closed off-channel habitat areas (oxbow lakes and sloughs).

8.5.4 Increased summer base flow

Flow regulation on the Sacramento River has increased summer base flow levels and may have, as a result, accelerated rates of progressive migration. Flow regulation of the Sacramento River has increased summer base flows to satisfy downstream irrigation demands while reducing the magnitude of winter flood peaks to satisfy to dam storage requirements and to mitigate downstream effects of flood flows (Figures 2-4 through 2-7) (CDWR 1980, Buer 1984, Lowney and Greco 2003). The estimated increase in summer base flows is on the order of 80–100% above pre-dam conditions during July and August (Kondolf et al. 2000, CDWR 2001, Snowden 2002, TNC 2003). While flows below bankfull do not impact rates of cutoff or off-channel habitat creation directly, they do influence progressive migration rates and patterns. In this way they can affect off-channel habitats indirectly by serving as an important control on channel planform and the resultant risk of cutoff when overbank flows occur.

Rates of progressive meander migration are assumed to reach a maximum on the falling limb of the hydrograph following periods of bankfull (or higher) flow (when shear stresses acting at the bank toe reach have reached a peak and falling water levels create positive pore pressures that accelerate bank failure), although progressive bank erosion has been observed at relatively low flows (Thorne 1992). Field work assessing bank erosion response to variable flows on the Sacramento River has shown that bank erosion may be initiated at flows as low as 10,600 cfs (Larsen et al., unpublished).

There is the potential for augmented summer baseflows to cause extended periods of accelerated bank erosion. This process, in combination with increased floodplain erodibility caused by clearing of riparian forest from the floodplain, could contribute an overall tendency towards channel straightening that would reduce the future pool of plausible chute cutoffs and thus reduce the rate of off-channel habitat formation.

Increased summer baseflows may have affected the elevation range of western pond turtles nesting areas. This may have had both direct and indirect effects on western pond turtle breeding and overall survival. Increases in summer baseflows may inundate historically significant nesting sites, but this is difficult to verify because there it is no data available on relative elevations of western pond turtle nest sites. In addition, current flow management practices have increased the number of flow reversals during summer months (CDWR 2001). Under historical conditions, there was probably little chance that nest sites selected in June would be inundated (and higher risks of egg mortality) later that summer. Under current conditions, this may no longer be true, at least for nest sites with low relative elevations, if they are located just above water surface elevation in June or July. Given that western pond turtles exhibit a high degree of site fidelity at breeding sites, eggs deposited at lower elevation sites may experience more frequent inundation under current conditions. Indirect effects may include a reduction in nesting habitat and increased chance of predation or road mortality (with higher baseflows forcing western pond turtles to travel further in order to reach suitable nesting areas).

8.5.5 Effects of bank revetment and levee construction

Bank revetment and levee construction have reduced the total floodplain area that can be reworked by the natural fluvial geomorphic process of channel migration, both via progressive meander migration and cutoff. These actions, combined with land conversion for agriculture and other human uses (Section 8.5.1), have resulted in a pronounced decrease in the total amount of floodplain habitat compared to historical conditions. This has undoubtedly reduced the abundance of off-channel aquatic habitats available to western pond turtles in oxbow lakes and sloughs. Bank armoring and levee construction, in conjunction with the effects of modification of flow and sediment regimes (as described above), have likely altered both the hydrologic connectivity of these off-channel aquatic habitats to the main channel and the terrestrialization process depicted in Figure 8-2.

As described in Chapter 7, land owners had begun implementing localized riprap style bank protection projects by as early as the mid to late 1800s, but the vast majority of bank protection structures were installed by the Army Corps from the mid 1960s through the 1980s, when an estimated 34% of the channel between RM 194–143.5 was covered in riprap or concrete (Greco et al., unpublished [2006a]), based on USACE 1986 and CDWR 2005). Overall, an estimated 48% of the channel from Red Bluff to Colusa (RM 243–143) is now covered by riprap on at least one margin (Larsen and Greco 2002; S. Greco, personal communication, 2006). Figure 7-12 provides an example of the relative amount and distribution of riprap and other bank armoring currently found in the reach between RM 229–218.

8.5.6 Reduced sinuousity and off-channel habitat complexity

As discussed in Chapter 3, conversion of riparian floodplain forest to agricultural uses has apparently led to reduced sinuousity in new cutoff bends (Table 3-9; Figure 3-10). This is presumably contributing to a decrease in the complexity of off-channel habitats. The combination of higher bank erodibility and lower bank roughness due to forest clearing presumably helps make banks in floodplains easier to erode such that cutoffs occur more frequently, resulting in a less sinuous channel. This has presumably been exacerbated by an increase in the relative frequency of neck cutoffs, which may be shorter and less complex than complete cutoffs. If the decreasing trend in sinuosity of cutoff bends continues as a result of anthropogenic disturbance, then the off-channel habitat created in the future via lateral channel change will most likely be less complex and potentially less abundant.

8.5.7 Changes in water quality and water temperature

Non-point source pollution from adjacent land use has the potential to degrade the quality of off-channel habitats. For example, agricultural runoff may contain detrimentally high levels of suspended sediment, nutrients (from fertilizer), herbicides, and pesticides. The extent to which non-point source pollution has affected western pond turtle populations on the Sacramento River is unknown. The close linkage of western pond turtle life history to oxbow habitats suggests that changes in water quality could affect health and survival, either directly, through toxic effects, or indirectly, by changing the abundance and distribution of zooplankton (the key food source for hatchlings). The long lifespan of western pond turtles may increase the potential for bioaccumulation of toxins, with potential negative effects on health (D. Holland, pers. comm., 2005). However, western pond turtles appear to be fairly tolerant of low water quality, although there has been little research on the subject. The absence of literature on documented adverse water quality effects and the presence of apparently healthy western pond turtles in wastewater treatment ponds in the Central Valley (Germano and Bury 2001; J. Germano, pers. comm., 2005), suggest that water quality may not be a key limiting factor for western pond turtle survival on the Sacramento River.

Flow regulation on the Sacramento River typically results in reduced summer temperatures (reduced by 9 to 18ºF [5 to 10ºC] to an average of 54ºF [12ºC]) and increased winter temperatures in the main river (increased by 4 to 9ºF [2 to 5ºC], from about 45–46ºF to 50–54ºF [7–8ºC to 10–12ºC]) compared to historical conditions prior to construction of Shasta Dam (Chapter 4). Because air temperatures in the Central Valley are generally very warm during the spring and summer months, it is unlikely that lower maximum and minimum water temperatures in the mainstem of the Sacramento River are causing significant behavioral changes in western pond turtle populations. Most western pond turtles prefer oxbow and side channel habitats where water temperatures are likely to be elevated. If suitable oxbow/slough habitat were not available, decreased summer temperatures in the mainstem might increase the amount of basking time required for thermoregulation and result in reduced growth rates and smaller size at sexual maturity, similar to the effects on western pond turtles of cold hypolimnetic reservoir releases that have been documented in the Trinity River (Reese 1996, Reese and Welsh 1998a and b).

8.5.8 Biotic interactions with non-native and disturbance-associated species

The introduction of non-native species can be detrimental to native species assemblages. Of particular concern to the conservationof western popnd turtles are non-native red-ear sliders (*Trachemys scripta elegans*), a turtle which has been introduced throughout California largely by escaping from or being intentionally released by pet owners (Bettelheim 2005). Red-ear sliders grow to larger adults sizes than western pond turtles, which may provide sliders with a competitive advantage when competing against smaller native turtles for basking habitat, food, and nesting habitat (Spinks et al. 2003, Reese 1996, and Holland 1994). Studies completed by Spinks et al. observed significant reduction in western pond turtle use of optimal habitat when red-ear sliders were present (Spinks 2003). Moreover, red-ear sliders are a vector for an unidentified upper respiratory disease, which can be fatal for native western pond turtles (Holland 1994). Recent research by Campbell (2004) demonstrated that viable populations of red-eared sliders are now found in the Sacramento Valley in many of the same areas as the western pond turtle. Both turtles favor lentic waters and have similar diets, as well as sharing the need for aerial basking as a component of metabolism (Campbell 2004). Other species may have more indirect effects on western pond turtle habitat and food resources. Introduced warmwater fish species, particularly centrarchids, may compete with hatchlings for zooplankton and other invertebrate prey items.

A number of species may prey on one or more western pond turtle life stage. Raccoons can prey on western pond turtles during all life stages, and have been observed in higher densities in areas where western pond turtle populations were found (Hays 1999, Germano 2005). Other possible predators include largemouth bass and bullfrogs, both of which would be expected to target hatchlings in particular, and are found in increasing numbers throughout California (Bettelheim 2005). Bullfrogs have been observed feeding on both hatchlings and juveniles (Holland 1994, Moyle 1973).

Reduction in nesting habitat quality may increase the risk of nest failure for a number of reasons. Semi-suitable nesting habitat may exist in agricultural areas adjacent to the river corridor. However, utilizing this habitat may increase the chance of egg mortality from inundation (via irrigation) or predation. Abnormally high raccoon populations (which have been linked with fragmentation of habitat, facultative supplemental feeding from garbage, and increased edge habitat) may severely limit western pond turtle recruitment, particularly if raccoons have easy access to nesting areas (Holland 1994; D. Holland, pers. comm., 2005). Eggs and hatchlings are particularly vulnerable to predation (Reese 1996, Holland 1994).

8.6 Conceptual Model of Current Conditions

Under current conditions, nearly all of the 87,000 ha (214,000 ac) of potential habitat occurring historically in the flood basin wetlands has been lost, although some western pond turtle habitat remains in several National Wildlife Refuges (Germano and Bury 2001) and along canals associated with rice fields (Lechner and Wilson 2004).

Most of the historical wetland habitat located within the riparian zone has also been lost, although remnant wetland and lentic habitat can still be found in off-channel sites (Figure 8-4). Most of the remaining off-channel habitat occurs from Red Bluff to Colusa (RM 245–140) (Figure 8-5). Below Colusa, levees, bank protection, and agricultural development have eliminated the majority of suitable off-channel aquatic habitats. The subreaches with greatest development of off-channel lentic habitats (as shown in Figure 8-5) correlate quite well with the zones of highest meander migration rates during the past century (Micheli and Larsen, in preparation; Constantine et al. unpublished; Figures 3-7 and 3-8).

Flow regulation and hypolimnetic reservoir releases have reduced the already limited suitability of the river above Red Bluff (RM 302–245) for western pond turtles by reducing summer water temperatures by 9–18ºF (5–10ºC) (from estimated temperatures of 59 to 68+ºF [15 to 20+ºC] in most years, to a fairly consistent 50–54ºF (10–12ºC) at the Keswick gage station).

The great reduction in potential habitat suggests that western pond turtle populations in the Sacramento Basin have been greatly reduced from historical conditions, although probably not as severely as the presumed decline in the Tulare Basin population caused by the loss of the three large shallow lakes (i.e., Tulare, Buena Vista, and Kern lakes; Germano and Bury 2001; see The Bay Institute 1998 for discussion of the historical and current extent of wetland and lentic habitat in this region).

In addition to large scale loss of habitat, many other factors have contributed or continue to contribute to declines in western pond turtle populations: introduced predators (e.g., largemouth bass, bullfrogs), increased numbers of native predators (e.g., raccoons), competitors, disease, and reduced water quality.

8.7 Management Implications

8.7.1 Actions to consider

8.7.1.1 Maintain or enhance current levels of off-channel habitats from Red Bluff to Colusa

Western pond turtle populations along the Sacramento River are not well documented but are expected to be most abundant in Red Bluff to Colusa reach (RM 243–143). Long-term conservation and enhancement, if possible, of western pond turtle populations and their habitats in this reach are likely to be important factors in maintaining a viable metapopulation of this species in the Sacramento Valley.

Although much uncertainty exists, which could be greatly reduced by conducting some basic surveys and monitoring (see Section 8.7.2 below), there appears to be near unanimous consensus of western pond turtle experts knowledgeable about Central Valley populations that off-channel habitats such as oxbow lakes and sloughs provide the primary habitat for western pond turtles in the Sacramento River corridor. Current understanding of the processes driving creation and succession of such aquatic habitats is described above and in Chapter 3. There are clear linkages between flow management and the physical processes affecting off-channel habitats that have a direct bearing on habitat suitability for western pond turtles. Additional field and modeling studies, including studies currently in progress (the off-channel habitat study component of this project and on-going studies by California State University (CSU) researchers) should help refine our understanding of these linkages and help clarify potential management actions.

Over the long term, there is a need to balance the processes that create new cutoffs with processes affecting subsequent successional dynamics of associated habitats (see Figure 8-2), in order to maintain or increase the number and acreage of off-channel water bodies present in the future. This balance is affected by the interactions of variable flow and sediment load with geologic constraints and riparian vegetation, which govern the rate and location of progressive channel meander and the potential for chute cutoffs to occur.

In order to maintain or enhance the potential for future cutoff formation through channel migration, opportunities for land-based management activities such as setting back levees or removing rip rap should be considered. These are highly controversial discussions, however, stakeholders, managers, and resource agencies have demonstrated an ability to work together Sacramento River to find locations where actions such as these may provide multiple benefits (e.g., to improve ecosystem health and flood protection, Golet et al. 2006).

8.7.1.2 Manage flows during the primary summer nesting season to reduce risk of nest inundation

Variable flows also have the potential to affect egg and hatchling survival. Although recent water operations have not generally created large fluctuations during the primary incubation period (summer), even small scale fluctuations might alter soil moisture in nests via capillary action or inundation. It seems unlikely that western pond turtle females would nest at low enough relative elevations to be affected by such fluctuations, but field studies are needed to confirm this.

Landscape context is locally important in providing the required juxtaposition of aquatic habitat and terrestrial nesting sites. It is also important to understand landscape linkages at broader scales. Such linkages would provide longer range dispersal corridors and a means for recolonization if local populations go extinct.

8.7.2 Potential performance metrics

Periodic monitoring can be used to assess progress towards achieving management goals. Such monitoring should be focused on specific performance measures or "metrics" that relate to our conceptual models of process-habitat-biota linkages and predicted effects of management actions on these linkages. Some potential performance metrics related to process-based habitat restoration of off-channel habitats used by the western pond turtle are presented below, followed by some metrics that could be used to track the responses of turtle populations over time.

8.7.2.1 Physical process and habitat metrics

- Area of off-channel habitats over time (with expected changes to occur primarily on a decadal rather than annual time scale)
- Area available for channel meander migration (i.e., area within levees or geologic constraints) and area of floodplain reworked per unit time
- Timing and rate of flow ramps at various times of the year, and frequency and magnitude of flow reversals during the primary nesting and incubation periods
- Number and area of potential habitat nodes on tributaries and in lands outside of the riparian zone (this metric would be useful in assessing potential connectivity among western pond turtle populations).

8.7.2.2 Biological metrics

- Distribution and abundance of western pond turtle, by aquatic habitat type, along the middle Sacramento River corridor
- Density and demographics (e.g., sex ratio, age-specific survival and growth rates) of turtles in a subset of representative sites
- Clutch size and nest success rates of western pond turtle nests in different upland habitat types.

8.7.3 Key hypotheses and uncertainties

There is a clear need to improve our understanding of western pond turtle distribution and abundance in the Sacramento River corridor. Existing uncertainties about western pond turtle populations and their local habitat needs greatly limits our ability to development effective management strategies to conserve this species of special concern. A one-time comprehensive survey of western pond turtle distribution, abundance, and habitat associations would allow us to test a number of the our hypotheses about factors currently limiting western pond turtle populations and the potential benefits versus costs of undertaking particular management actions.

A comprehensive baseline survey should include the following:

- A systematic survey of western pond turtle distribution and abundance from Keswick to the Delta. Ideally this would include a comprehensive survey of all potential western pond turtle aquatic habitats in the river corridor. However, access to private property, cost, and other logistical constraints might require adopting a less intensive effort that would focus on surveyed representative habitat units and sub-reaches.
- Survey techniques include visual searches, trapping and hand capture. Hand capture is not likely to be effective in off-channel habitats and visual searches may sometimes yield false negatives, therefore a combination of visual searches and more intensive trapping seems like the best approach.
- concurrent collection of habitat association data (e.g., aquatic habitat type, water depth and temperature, species composition and structure of aquatic and riparian vegetation, density of emergent basking sites, food abundance, potential predators or competitors) and creation of a GIS database for subsequent spatial analysis.
- Because hatchlings may not be adequately surveyed using methods appropriate for older juveniles and adults, focused surveys for hatchlings should be conducted in habitats identified as being potentially suitable for hatchling rearing (but including some surveys in other habitats to test our ability to accurately predict habitat suitability for hatchlings).
- Subsequent to the survey, a subset of habitat units and sub-reaches should be selected as representative "index" sites for periodic resurveying as part of a longer term trends monitoring effort.

In addition to the baseline survey described above, the following types of focused studies in the Sacramento River corridor would also be beneficial:

- Habitat and microhabitat associations for rearing of hatchlings and juveniles, including:
  - Types of habitats and microhabitats used
  - Water depth and temperature
  - Type and density of aquatic vegetation
  - Food availability and diet studies
  - Thermoregulatory behavior (use of aquatic and terrestrial basking sites)
- Radiotelemetry studies of a variety of age classes to look at the following:
  - Female nest site selection behavior and mapping of nest site locations with associated habitat information (relative elevation, distance from nearest water body, soil texture, slope, aspect, vegetation, etc.)
  - Year-round movement patterns of adults and juveniles to examine:
    - Local seasonal movement patterns in terrestrial and aquatic environments
    - Broader scale dispersal patterns in both terrestrial and aquatic environments

    - Impacts of flood events with varying magnitudes, timings and durations
- Phenology (timing) of nesting, incubation, hatching, and hatchling emergence and dispersal from the nest
- Investigate potential for compatible land uses adjacent to potential western pond turtle habitats. This would include examination of potential strategies for developing "wildlife-friendly" agriculture that would reduce impacts to local western pond turtle populations.

Studies are needed to assess the relative importance of channel and floodplain habitats along the middle Sacramento River, flood basin areas downstream of Colusa, habitats along the upper Sacramento River and in major tributaries:

- Assess habitat suitability of existing perennial ponds in the Yolo and Sutter bypasses to determine their potential to support western pond turtle populations in the bypasses.
- Identify existing and potential western pond turtle habitat in wildlife refuges.
- Follow up on work conducted by Lechner and Wilson (2004) in areas used for rice farming.
- Examine landscape connectivity of existing habitats or populations, particularly connectivity with the Sacramento River corridor.
- Identify potential habitat nodes to enhance from Keswick to Red Bluff:
    a. Identify existing and potential western pond turtle habitat along the river corridor
    b. Conduct baseline survey of western pond turtle distribution and abundance in the reach
- Conduct baseline survey of western pond turtle distribution and abundance in the tributaries

A long-term effort to develop a western pond turtle population/metapopulation dynamics model for the Sacramento Valley would facilitate conservation planning:

- Determine data needed for modeling and interact with baseline survey and trend monitoring program, and the recommended focused studies described above, to facilitate collection of necessary data.

8.8 References

Ashton, D. T., A. J. Lind, and K. E. Schlick. 1997. Western pond turtle (*Clemmys marmorata*). Natural history. USDA Forest Service, Pacific Southwest Research Station, Redwood Sciences Laboratory, Arcata, California.

Ashton, D. T. 2005. Influence of altered thermal regime on body size and age of maturation on western pond turtles (*Clemmys marmorata*), Trinity County, California. Western pond turtle workshop: ecology and conservation. The Wildlife Society, San Francisco Bay Area Chapter.

Bay Institute, The (TBI). 1998. From the Sierra to the sea: the ecological history of the San Francisco Bay-Delta watershed. San Rafael, California.
http://www.bay.org/sierra_to_the_sea.htm

Bettaso, J. 2005. Basking patterns and thermal regulatory behaviors of western pond turtles (*Clemmys marmorata*) between two thermal regimes in dammed and non-dammed forks of the Trinity River, Trinity County, California. Western pond turtle workshop: ecology and conservation. The Wildlife Society, San Francisco Bay Area Chapter.

Bettelheim, M. P. 2005. The western pond turtle, *Clemmys marmorata*: a natural history of the species. Walnut Creek, California.

Buer, K. 1984. Middle Sacramento River spawning gravel study. California Department of Water Resources, Northern District, Red Bluff.

Bury, R. B. 1972. Habits and home range of the Pacific pond turtle, *Clemys marmorata*, in a stream community. Doctoral dissertation. University of California, Berkeley.

Bury, R. B. 2005. A 40-yr chelonian odyssey with the western pond turtle: what the heck is the status of its populations, ecology and conservation? Western pond turtle workshop: ecology and conservation. The Wildlife Society, San Francisco Bay Area Chapter.

Buskirk, J. R. 2002. The western pond turtle, *Emys marmorata.* Radiata 11: 3-30.
http://pondturtle.com/Buskirk,%20James%20R.%202002.pdf

Campbell, J. 2004. Interspecific competition in basking turtles or is California's *Clemmys marmorata* competing with the invasive *Trachemys scripta* for available resources? Abstract of presentation at the annual meeting of the American Society of Ichthyologists and Herpetologists, Norman, Oklahoma, 26-31 May 2004.
http://www.asih.org/meetings/2004/adams_to_echelle.pdf.

CDWR (California Department of Water Resources). 1980. Upper Sacramento River spawning gravel study. Report. Prepared for California Department of Fish and Game by CDWR, Northern District, Red Bluff.

CDWR (California Department of Water Resources). 2001. RE: Transmittal of Indicators of Hydrologic Alteration data files on CD-ROM. Memorandum to Glenn Pearson from Stacy Cepello and Shawn Pike, CDWR, Sacramento, California. 23 February.

Collins, J. N., K. G. Gallagher, and V. H. Resh. 1985. Thermal characteristics of aquatic habitats at Coyote Hills Marsh: implications for simulation and control of *Anopheles* mosquitoes. Proceedings of the California Mosquito and Vector Control Association 53:83-86.

Constantine, C., T. Dunne, and M. Singer. Unpublished. Controls on spatial differences in meander migration rates in a large gravel-bed river. Submitted to Earth Surface Processes and Landforms.

Cook, D. G., and J. Martini-Lamb. 2004. Distribution and habitat use of Pacific pond turtles in a summer impounded river. Transactions of the Western Section of the Wildlife Society 40: 84-89.

Ewert, M. A., D. R. Jackson, and C. E. Nelson. 1994. Patterns of temperature-dependent sex determination in turtles. Journal of Experimental Zoology 270: 3-15.

Feyrer, F., T. R. Sommer, S. C. Zeug, G. O'Leary, and W. Harrell. 2004. Fish assemblages of perennial floodplain ponds of the Sacramento River, California (USA), with implications for the conservation of native fishes. Fisheries Management and Ecology 11: 335-344.

Germano, D. 2005. Western pond turtles from north to south: what do we know? Western pond turtle workshop: ecology and conservation. The Wildlife Society, San Francisco Bay Area Chapter.

Germano, D. J., and R. B. Bury. 2001. Western pond turtles (*Clemmys marmorata*) in the Central Valley of California: status and population structure. Transactions of the Western Section of the Wildlife Society 37: 22-36.

Gibbs, J. P., and D. A. Steen. 2005. Historical trends in turtle sex ratios in the United States: long-term implications of road mortality. Conservation Biology 19: 552-556.

Golet, G. H., D. L. Brown, E. E. Crone, G. R. Geupel, S. E. Greco, K. D. Holl, D. E. Jukkola, G. M. Kondolf, E. W. Larsen, F. K. Ligon, R. A. Luster, M. P. Marchetti, N. Nur, B. K. Orr, D. R. Peterson, M. E. Power, W. E. Rainey, M. D. Roberts, J. G. Silveira, S. L. Small, J. C. Vick, D. S. Wilson, and D. M. Wood. 2003. Using science to evaluate restoration efforts and ecosystem health on the Sacramento River Project, California. Pages 368-385 *in* P. M. Faber, editor. California riparian systems: processes and floodplain management, ecology, and restoration. 2001 Riparian habitat and floodplain conference proceedings. Riparian Habitat Joint Venture, Sacramento, California.

Golet G. H., M. D. Roberts, E. W. Larsen, R. A. Luster, R. Unger, G. Werner, and G. G. White. 2006. Assessing societal impacts when planning restoration on large alluvial rivers: A case study of the Sacramento River Project, California. Environmental Management 37:862-879.

Greco, S. E. 1999. Monitoring riparian landscape change and modeling habitat dynamics of the yellow-billed cuckoo on the Sacramento River, California. Doctoral dissertation. University of California, Davis.

Greco, S. E., A. K. Fremier, E. W. Larsen, and R. E. Plant. Unpublished (2006a). Tracking surficial patterns of floodplain chronology on a large meandering river: analysis of land production rates and riparian vegetation distribution over land age gradients. Submitted to Ecography.

Greco, S. E., and R. E. Plant. 2003. Temporal mapping of riparian landscape change on the Sacramento River, miles 196-218, California, USA. Landscape Research 28: 405-426.

Hays, D. W., K. R. McAllister, S. A. Richardson, and D. W. Stinson. 1999. Washington State recovery plan for the western pond turtle. Washington Department of Fish and Wildlife, Olympia.

Holland, D. C. 1989. Synopsis of the ecology of the western pond turtle, *Clemmys marmorata*. Prepared for U.S. Fish and Wildlife Service, National Ecology Research Center, Fort Collins, Colorado.

Holland, D. C. 1994. The western pond turtle: habitat and history. Final Report DOE/BP-62137-1. Bonneville Power Administration, Portland, Oregon.

Holland, D. C., and B. Bury. In press. *Clemmys marmorata* (Baird and Girard 1852) western pond turtle. *In* P. C. Pritchard and A. G. Rhodin, editors. Conservation biology of freshwater turtles. Chelonian Research Monograph Volume II.

Jennings, M. R., M. P. Hayes, and D. C. Holland. 1992. A petition to the U.S. Fish and Wildlife Service to place the California red-legged frog (*Rana aurora draytonii*) and the western pond turtle (*Clemmys marmorata*) on the list of endangered and threatened wildlife and plants. Letter to M. Plenert, Regional Director, U.S. Fish and Wildlife Service, Region 1, Portland, Oregon. January 15.

Jennings, M. R., and M. P. Hayes. 1994. Amphibian and reptile species of special concern in California. Final Report. Prepared by California Academy of Sciences, San Francisco and Portland State University, Oregon for California Department of Fish and Game, Inland Fisheries Division, Rancho Cordova.

Katibah, E. F. 1984. A brief history of riparian forests in the Central Valley of California. Pages 23-29 *in* R. E. Warner and K. M. Hendrix, editors. California riparian systems: ecology, conservation, and productive management. University of California Press, Berkeley.

Kondolf, G. M., T. Griggs, E. W. Larsen, S. McBain, M. Tompkins, J. G. Williams, and J. Vick. 2000. Flow regime requirements for habitat restoration along the Sacramento River between Colusa and Red Bluff. CALFED Bay Delta Program, Integrated Storage Investigation, Sacramento, California.

Larsen, E. W., and S. E. Greco. 2002. Modeling channel management impacts on river migration: a case study of Woodson Bridge State Recreation Area, Sacramento River, California, USA. Environmental Management 30: 209-224.

Larsen, E. W., A. K. Fremier, and S. E. Greco. Unpublished Cumulative effective stream power and river channel migration on the Sacramento River, California, USA. *Submitted to Environmental Management.*

Lechner, G., and D. S. Wilson. 2004. Activity patterns, habitat use and population characteristics of the western pond turtle inhabiting rice agriculture. Abstract of presentation at the annual meeting of the American Society of Ichthyologists and Herpetologists, Norman, Oklahoma, 26-31 May 2004. http://www.asih.org/meetings/2004/laduc_to_roberts.pdf.

Lovich, J. 1999. Western pond turtle (*Clemmys marmorata*). U.S. Geological Survey, Western Ecological Research Center, Department of Biology, University of California, Riverside. http://www.ca.blm.gov/pdfs/cdd_pdfs/clemmys1.pdf.

Lowney, C. L., and S. E. Greco. 2003. Flood frequency analysis of the Sacramento River at Bend Bridge. Technical Memorandum. Prepared for California Department of Water Resources, Northern District, Red Bluff by Landscape Analysis and Systems Research Laboratory, Department of Environmental Design, University of California, Davis.

Micheli, E. R., J. W. Kirchner, and E. W. Larsen. 2004. Quantifying the effect of riparian forest versus agricultural vegetation on river meander rates, central Sacramento River, California, USA. River Research and Applications 20: 537-548.

Micheli, E. R., and E. W. Larsen. In preparation. River channel cutoff dynamics, Sacramento River, California, USA.

Morken, I., and G. M. Kondolf. 2003. Evolution assessment and conservation strategies for Sacramento River oxbow habitats. Prepared for The Nature Conservancy, Sacramento River Project. Berkeley, California.

Moyle, P. B. 1973. Effects of introduced bullfrogs, *Rana catesbeiana*, on the native frogs of San Joaquin Valley, California. Copeia 1973: 18-22.

Nussbaum, R. A., Jr. E. D. Brodie, and R. M. Storm. 1983. Amphibians and reptiles of the Pacific Northwest. University Press, Moscow, Idaho.

Parham, J. F., and C. R. Feldman. 2002. Generic revisions of emydine turtles. Turtle and Tortoise Newsletter 6: 28-30.

Piegay, H., A. Citterio, and L. Astrade. 1998. Ligne de débris ligneux et recoupement de méandres, exemple du site de Mollon sur l'Ain (France). Zeitzchrift für Geomorphologie 42: 187-208.

Rathbun, G. B., N. Siepel, and D. C. Holland. 1992. Nesting behavior and movements of western pond turtles (*Clemmys marmorata*). The Southwestern Naturalist 37: 319-324.

Reese, D. A. 1996. Comparative demography and habitat use of western pond turtle in northern California: the effects of damming and related alterations. Doctoral dissertation. University of California, Berkeley.

Reese, D. A., and H. H. Welsh, Jr. 1997. Use of terrestrial habitat by western pond turtles, *Clemmys marmorata*: implications for management. Pages 352-357 *in* Proceedings: Conservation, restoration, and management of tortoises and turtles. An international conference. New York Turtle and Tortoise Society.

Reese, D. A., and H. H. Welsh, Jr. 1998a. Comparative demography of *Clemmys marmorata* populations in the Trinity River of California in the context of dam-induced alterations. Journal of Herpetology 32: 505-515.

Reese, D. A., and H. H. Welsh, Jr. 1998b. Habitat use by western pond turtles in the Trinity River, California. Journal of Wildlife Management 62: 842-853.

Shaffer, H. B. 2005. Survival of pond turtles in modified waterways: how can it work, and why does it matter? Western pond turtle workshop: ecology and conservation. The Wildlife Society, San Francisco Bay Area Chapter.

Shaffer, H. B., P. Meylan, and M. L. McKnight. 1997. Tests of turtle phylogeny: molecular, morphological, and paleontological approaches. Systematic Biology 46: 235-268.

Singer, M. B., and T. Dunne. 2001. Identifying eroding and depositional reaches of valley by analysis of suspended sediment transport in the Sacramento River, California. Water Resources Research 37: 3371-3381.

Snowden, V. L. 2002. Hydrologic alterations to the Sacramento River and the effects on cottonwood seedlings. Master's thesis. California State University, Chico.

Spinks, P. 2005. Rangewide molecular analysis of the western pond turtle (*Emys marmorata*): cryptic variation, isolation by distance, and their conservation implications. Western pond turtle workshop: ecology and conservation. The Wildlife Society, San Francisco Bay Area Chapter.

Spinks, P. Q., G. B. Pauly, J. J. Crayon, and H. B. Shaffer. 2003. Survival of the western pond turtle (*Emys marmorata*) in an urban California environment. Biological Conservation 113: 257-267.

Stebbins, R. C. 2003. A field guide to western reptiles and amphibians. Third edition. Houghton Mifflin Company, Boston-New York.

Teisseyre, A. 1977. Meander degeneration in bed-load proximal streams: repeated chute cutoff due to barhead gravel accretion--a hypothesis. Geologia Sudetica 12: 103-115.

Thorne, C. R. 1992. Bend scour and bank erosion on the meandering Red River, Louisiana. Pages 95-115 *in* P. Carling and G. Petts, editors. Lowland floodplain rivers: geomorphological perspectives. Wiley and Sons, Chichester, England.

TNC (The Nature Conservancy). 2003. Management and monitoring recommendations for floodplain habitats of the Chico Landing Subreach (RM 178-206), Sacramento River, California. The Nature Conservancy, Chico, California.

USFWS (U.S. Fish and Wildlife Service). 1992. Endangered and threatened wildlife and plants; 90-day finding and commencement of status reviews for a petition to list and western pond turtle and California red-legged frog. Federal Register 57: 45761-45762.

Zeiner, D. C., W. F. Laudenslayer Jr., and K. E. Mayer, editors. 1988. California's wildlife. Volume I. Amphibians and reptiles. California Statewide Habitat Relationships System. California Department of Fish and Game, Sacramento.

Figures

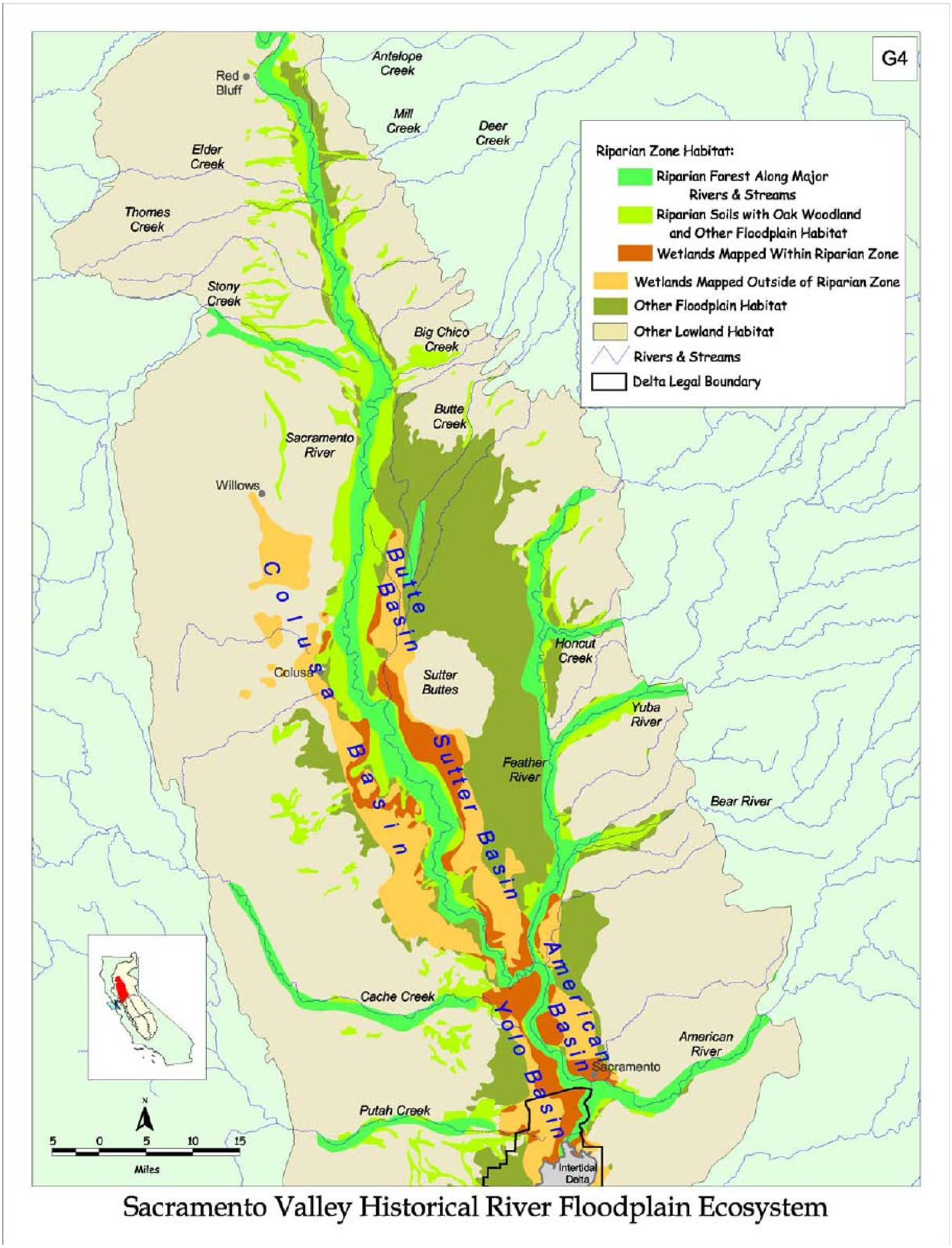


Figure 8-1. Estimated historical extent of riparian forests, woodlands, and wetlands, and flood basin wetlands in the Sacramento Valley (from The Bay Institute 1998).

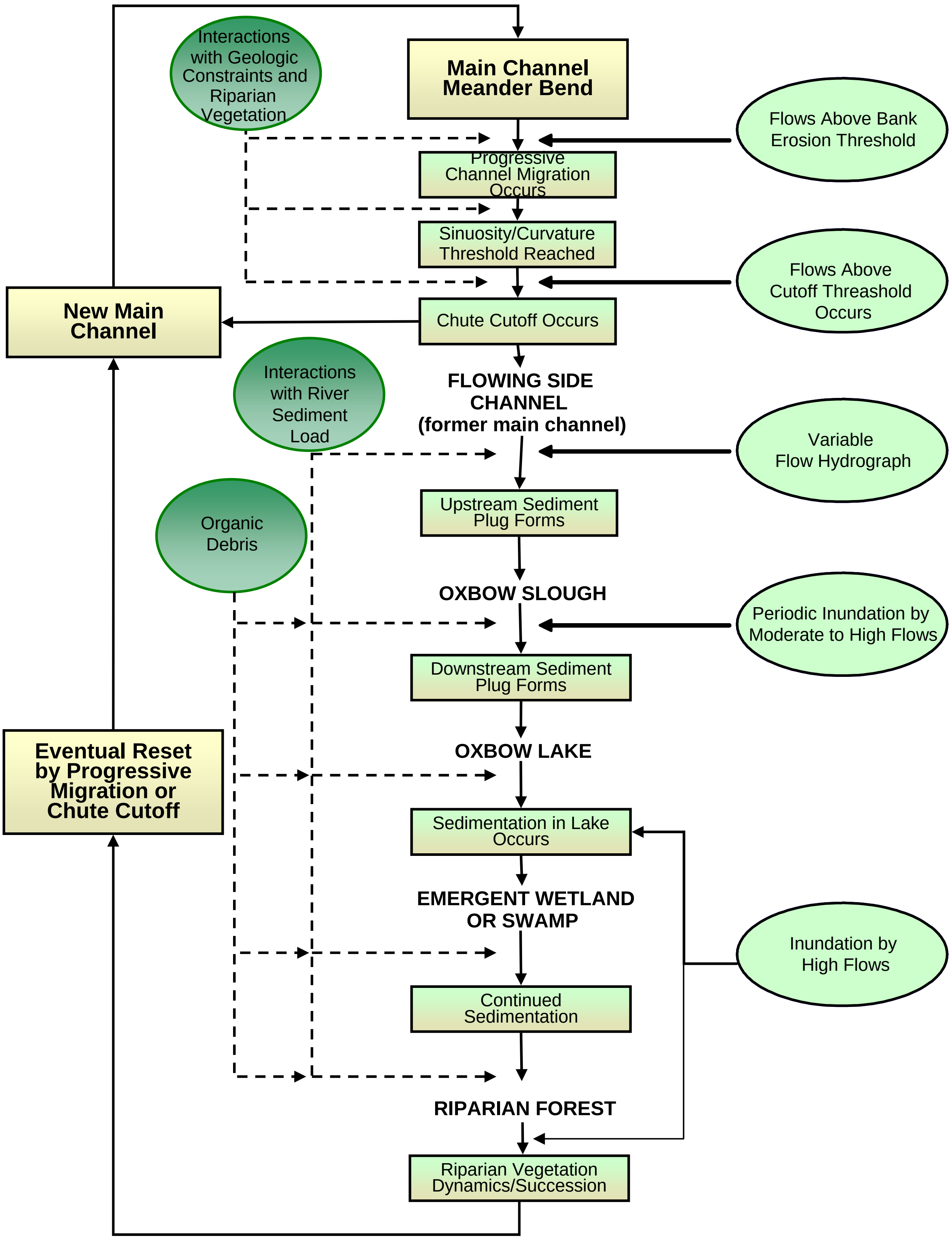


Figure 8-2. Habitats associated with chute cutoffs.

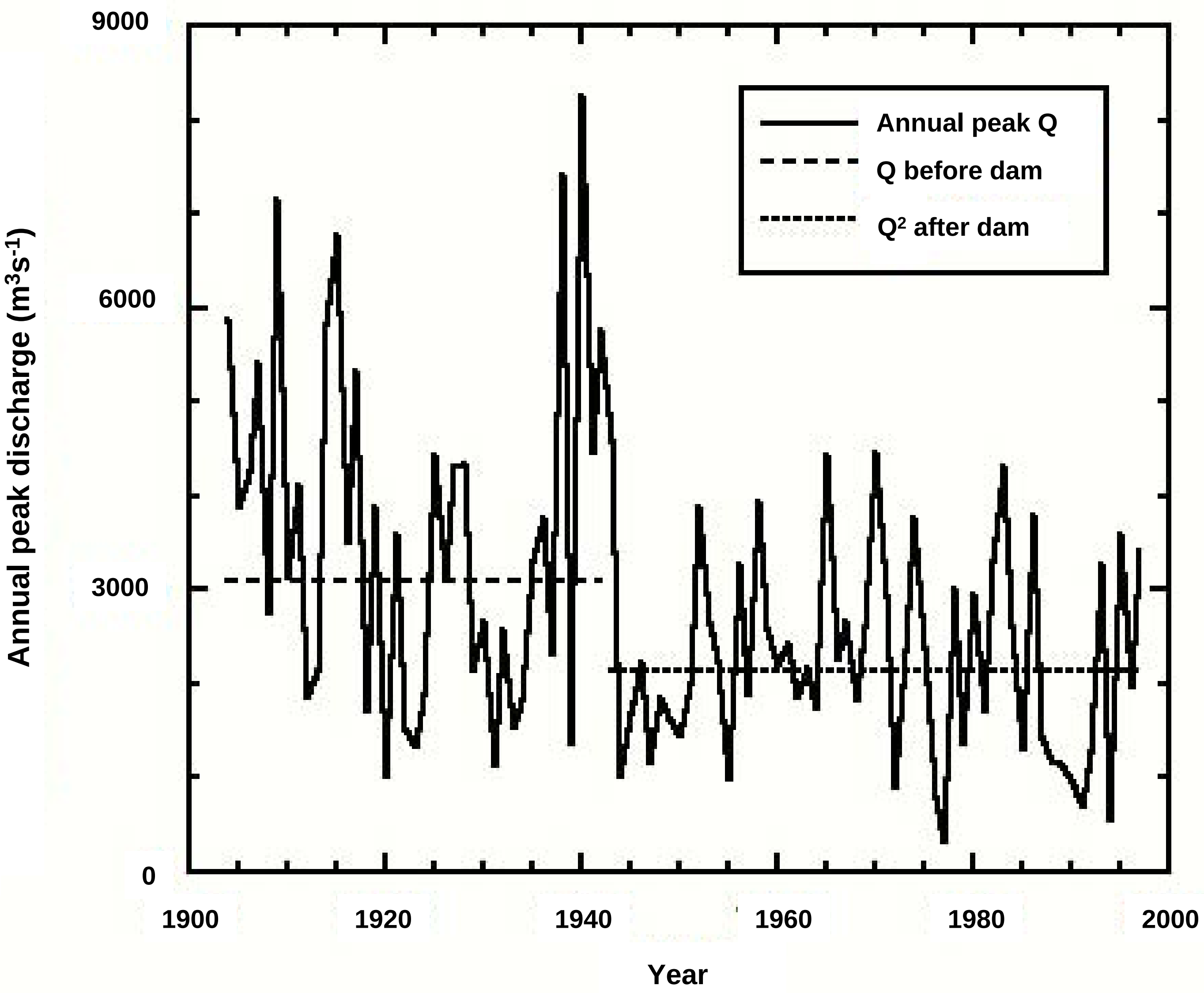


Figure 8-3. Peak flows at Bend Bridge Gage, 1904-2000 (source: Larsen et al. in review).

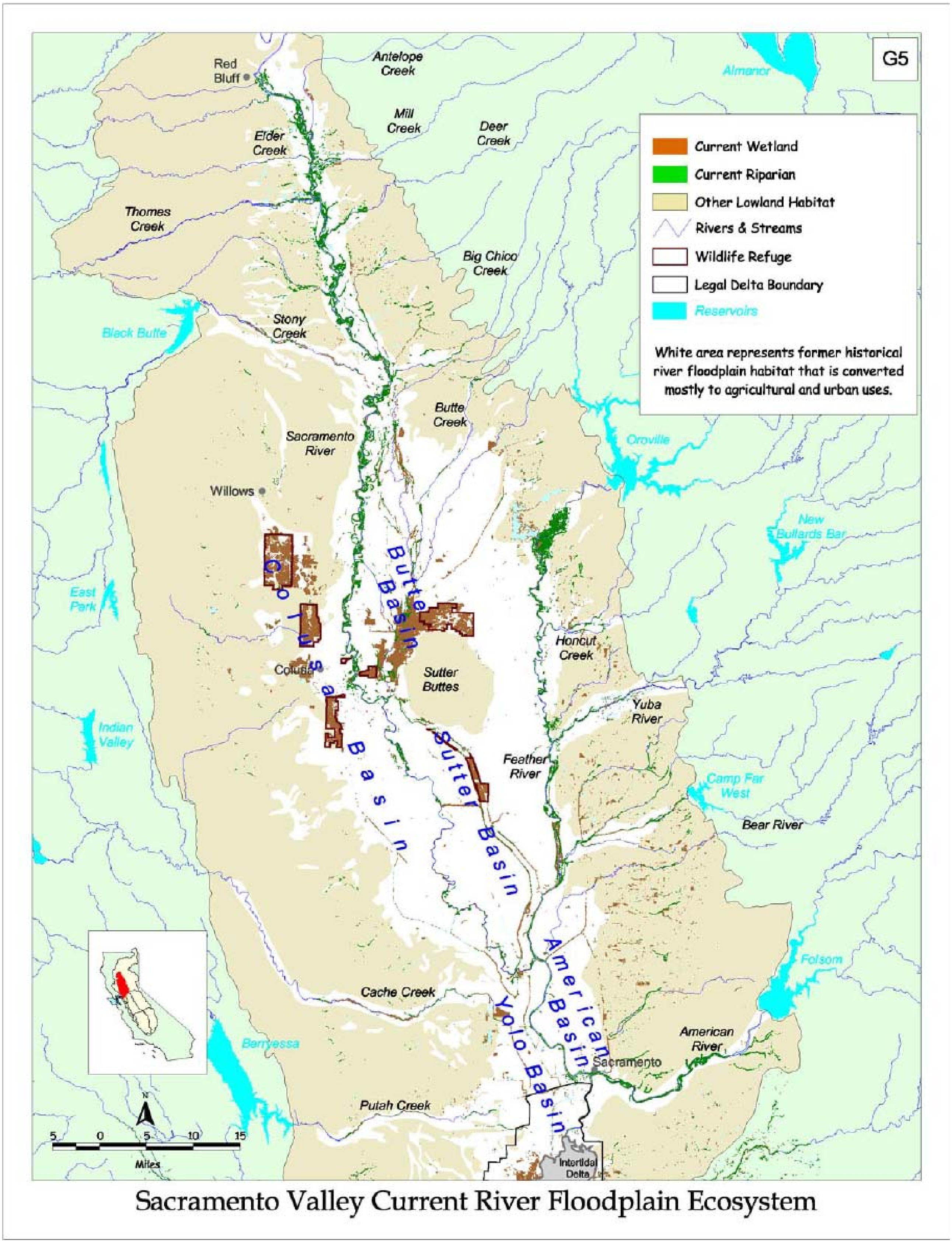


Figure 8-4. Current extent of riparian forests and riparian and flood basin wetlands in the Sacramento Valley (from The Bay Institute 1998).

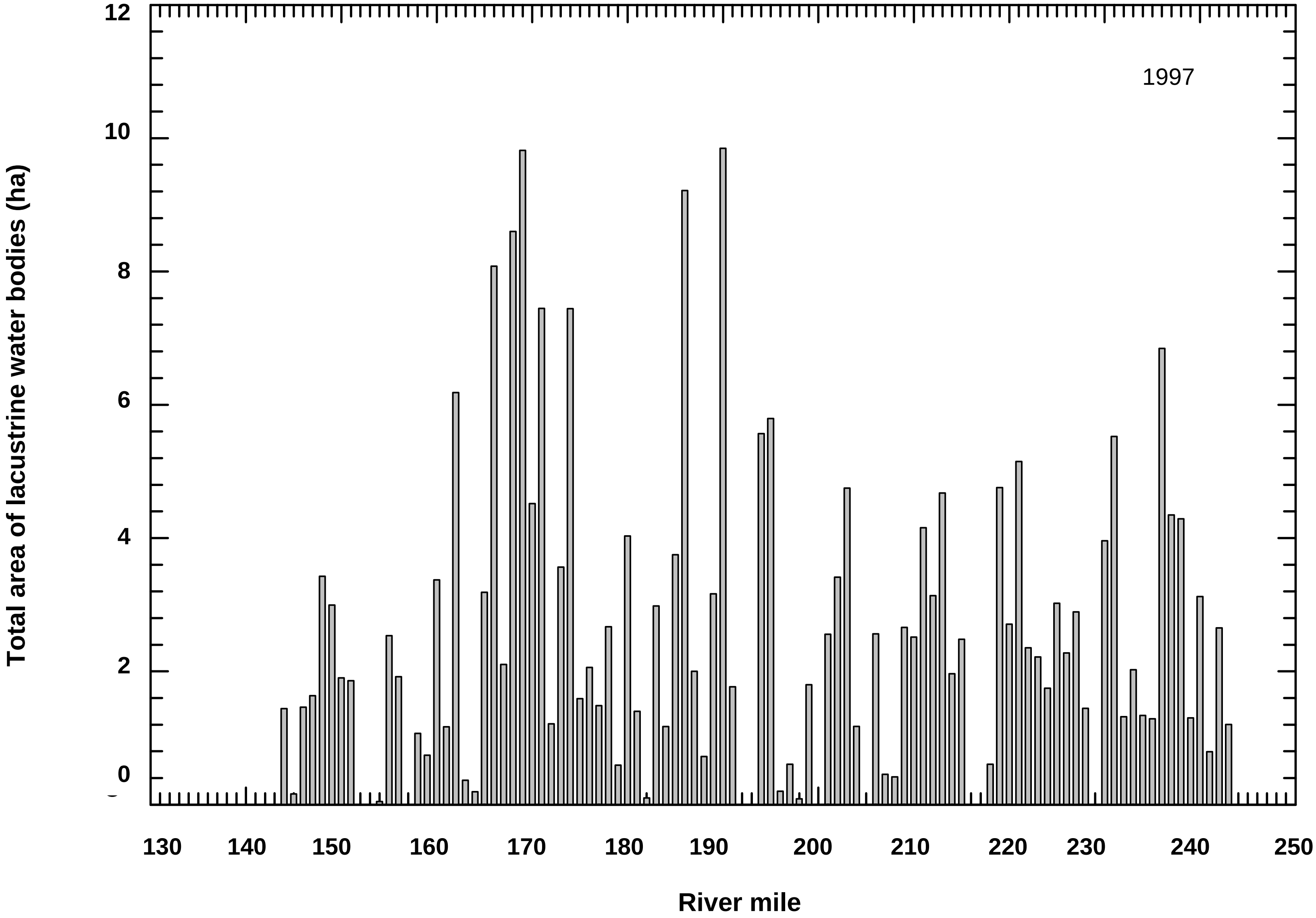


Figure 8-5. Distribution of off-channel lacustrine waterbodies, by river mile, based on mapping from aerial photograph analysis conducted by Greco and Alford (2003).

9 FREMONT COTTONWOOD

In the Central Valley of California, Fremont cottonwood (*Populus fremontii* ssp. *fremontii*) is the tree species that dominates the near-river forests. It is the largest and fastest-growing tree to colonize young floodplain surfaces. Soon after establishment, it provides ecological structure to the riparian ecosystem by stabilizing the substrate, fixing carbon, generating large woody debris, and creating vertical stratification for wildlife habitat. Cottonwoods have a suite of life history traits that allow them to establish and thrive under a highly variable natural flow regime and a spatially complex geomorphic setting. Modification of the natural flow regime by damming and diversion has been shown to substantially reduce the rate of regeneration of new cottonwood forests in many regulated rivers in western North America (e.g., Rood and Mahoney 1990, Braatne et al. 1996, Rood et al. 2003, Rood et al. 2005).

Fremont cottonwood was selected as a focal species primarily because (1) it is known to be sensitive to alterations in the natural flow regime in rivers of Western North America, and (2) cottonwood trees play a pivotal role in the establishment and dynamics of riparian forests and the creation of habitat for riparian-dependent wildlife (see Section 1.5 for more details of the selection process for this focal species).

9.1 Riparian Zone Definition and Ecological Values

Riparian zones are the three-dimensional interface between terrestrial and aquatic ecosystems (Gregory et al. 1991), with complex, dynamic landscapes that are structured in large part by effects of climate, topography, hydrology, and geomorphology (Gregory et al. 1991, Naiman and Decamps 1997, NRC 2002, Naiman et al. 2005). Beginning in the 1970s, riparian ecosystems in California and the western United States have become increasingly recognized both for their ecological value and for the severity of their loss in many areas due to effects of alterations in hydrology (caused by dams and diversions), conversion of near channel environments for agricultural and urban uses, and placement of bank hardening materials (e.g., riprap) to reduce bank erosion. Additionally, construction of levees has disconnected large portions of the floodplain from the river.

Riparian zones are now widely recognized as centers of biodiversity and corridors for dispersal of plants and animals across the landscape (Gregory et al. 1991, Johansson et al. 1996). They filter nutrients and agricultural chemicals from runoff, stabilize channel banks, and provide leaf litter to aquatic food webs, large woody debris and overhead cover for fish, nesting habitat and migratory corridors for terrestrial wildlife, and provide aesthetic value to the landscape (Mitsch and Gosselink 1993, Malanson 1993, Naiman and Descamps 1997, NRC 2002, Naiman et al. 2005).

Riparian zones in the Central Valley are threatened by land conversion (including clearing of riparian forests, bank hardening, and levee construction), flow alteration, climate change, and introduction of invasive non-native species. Widespread agricultural development in the 19th and 20th centuries reduced the area of native riparian vegetation to 5–10 percent of its pre-settlement extent (Katibah 1984). Flow regulation and levee construction on nearly all major rivers and streams, including the Sacramento River, have created a less dynamic physical regime with generally reduced variability in the flow regime, which alters the ecological processes that generate and maintain the natural, dynamic ecosystem (Ligon et al. 1995, Poff et al. 1997, Lytle and Poff 2004). Furthermore, the projected regional changes in climate due to global warming promise to influence ecological processes in complex ways.

9.2 Fremont Cottonwood (*Populus fremontii*)

9.2.1 Taxonomy

Fremont cottonwood (*Populus fremontii* ssp. *fremontii*) belongs to the section *Aigeiros* within the genus *Populus* (Eckenwalder 1996). When populations occur in sympatry, Fremont cottonwood is known to form natural hybrids with other members of section *Aigeiros* and with black cottonwood (*P. balsamifera* ssp. *trichocarpa*) and narrowleaf cottonwood (*P. angustifolia*), both of which belong to the section *Tacamahaca* (Eckenwalder 1996). Fremont cottonwood does not hybridize with more distantly related members of the genus, including quaking aspen (*P. tremuloides*) (Eckenwalder 1996). Cottonwoods and willows (*Salix* spp.) are the only temperate trees with the combination of female catkins—seeds that are borne on parietal placentas in thin-walled capsules—and a coma (a tuft) of cottony hairs on the seed that aids in wind dispersal (Eckenwalder 1996). Both cottonwoods and willows are members of the family Salicaceae (Hickman 1993, Eckenwalder 1996).

9.2.2 Geographic distribution and population trends

9.2.2.1 General distribution

The range of Fremont cottonwood includes the valleys and lower foothills of the southern and middle Rocky Mountains, the Great Basin, Mexico, and central and southern California (Stuart and Sawyer 2001). In California, Fremont cottonwoods are abundant in riparian areas that have a perennial water supply at or near the ground surface, with a range that includes the Central Valley, the foothills of the Sierra Nevada, the coast ranges, and the Mojave and Colorado deserts, at elevations ranging from near sea level to between 3,000 ft (915 m) and 5,000 ft (1,525 m) (Stuart and Sawyer 2001, Griffin and Critchfield 1972, Sudworth 1908).

Cottonwood forests (a multi-layered riparian forest type that includes Fremont cottonwood as the dominant overstory-forming species) are found along the banks of the middle Sacramento River and its floodplains from the Delta to Keswick Dam (Vaghti and Greco in press, Greco et al in review, Vaghti 2003, SRCAF 2003, Greco 1999, Katibah 1984). The greatest extent of cottonwood forest remaining along the Sacramento River occurs between Red Bluff and Colusa (Vaghti and Greco, in press, SRCAF 2003) (also see Figures 8-1 and 8-4 in Chapter 8 for a comparison of current and historical extent of riparian forests along the Sacramento River).

9.2.2.2 Population trends

Riparian cottonwood forests, once abundant throughout much of the southwestern United States, have been reduced by 70 to 95% or more, with only small remnants remaining in many areas (Johnson and Haight 1984, National Research Council 1992, Braatne et al 1996). The Sacramento Valley has experienced even greater declines, with loss of about 98.5% of riparian forests between 1850 and the present (Katibah 1984, Greco 1999). The meandering 100-mile (160 km) section of the middle Sacramento River from Red Bluff to Colusa has a considerably larger percentage of its historical riparian zone remaining (~10%) than other reaches, as determined by comparing Greco's 1999 rendition of the Holmes 1913 soils map with CSU Chico's 1999 mapping effort (Golet et al. 2003).

9.2.3 Life history and habitat requirements

Fremont cottonwoods are dioecious, drought- and shade-intolerant trees that grow along stream channels or riparian corridors. The life history and ecology of Fremont cottonwood are closely linked to river system patterns and processes. Fremont cottonwoods, like other riparian, pioneer tree species, have evolved life history traits that depend on natural fluvial processes, particularly spring snowmelt floods. Fremont cottonwoods are highly dependent on soil moisture and occur in alluvial stream channels and riparian corridors where subsurface water remains within several meters of the surface throughout the growing season.

Fremont cottonwoods reach reproductive maturity after five to 10 years (Reichenbacher 1984, as cited in Braatne et al. 1996). Male and female trees flower between February and March and small, cotton-borne seeds are dispersed by air or water between March and April (Reichenbacher 1984, Asplund and Gooch 1988, Stromberg et al. 1991, all as cited in Braatne et al. 1996). This time frame approximately coincides with the recession of historical spring snowmelt floods (Table 9-1 and Figure 9-1). Fremont cottonwoods can also reproduce asexually, primarily through tree fall and crown breakage during floods or storms, although vegetative reproduction is less common in Fremont cottonwoods and other members of section *Aigeiros* than it is with black and narrowleaf cottonwoods in section *Tacamahaca* (Braatne et al. 1996).

Seeds are generally viable for one to three weeks under natural conditions, and can germinate in 24 hours on bare, moist alluvial surfaces (Horton et al. 1960, as cited in Braatne et al. 1996; Stella 2005, Stillwater Sciences 2006, Stella et al. in press). Fremont cottonwood seeds require bare, moist sandy, humous, or gravelly soils for germination. Large numbers of germinating seeds and seedlings are often found on point bars and newly exposed alluvial substrates in late spring and early summer. During seedling development, the species requires adequate root contact with ground water. Peak seed release in *Populus fremontii* typically occurs during a 4–6 week period between March and June, with the specific timing varying among locations and years, at least partially in response to variations in local climate (Stella et al. in press). Along lower elevation desert rivers in Arizona seed release typically peaks in March or April (Braatne et al. 1996, Reichenbacher 1984), while in California's Central Valley peak seed release tends to occur sometime during mid-April through early July (Peterson 2002, Stillwater Sciences 2003, Stella 2005, Morgan 2005, Morgan and Henderson 2005b, Stillwater Sciences 2006, Stella et al in press).

First year seedlings of Fremont cottonwood tend to allocate more energy to root growth rather than stem growth (Stella 2005, Stillwater Sciences 2006). Similarly, during sapling development, energy is initially allocated more to root growth than to height growth (Braatne et al. 1996). After the root system is well established, older saplings and young trees grow quickly and can reach heights of 33–49 ft (10–15 m) by the time they are reproductively mature at ages of 5–10 years old (Cooper 1990, Van Haverbeke 1990, both as cited in Braatne et al. 1996). Fremont cottonwood can reach ages of 130 years or more (Shanfield 1983, Howe and Knopf 1991, Hunter et al. 1987, Szaro 1989, all as cited in Braatne et al. 1996, Sawyer and Keeler-Wolf in press).

Recruitment of new trees is hindered by reduced sediment supply and flood magnitude, and alteration of flood timing that have resulted from flow regulation. These changes in flow and sediment supply reduce the creation of suitable substrates for cottonwood seedling germination. While Fremont cottonwood seedlings are tolerant of being inundated for extended periods and siltation, seedlings that germinate on low depositional bars are often scoured away by subsequent regulated winter flows. Seedlings that manage to establish on higher floodplain sites (where the risk of subsequent scour is reduced) often die of desiccation because their root growth cannot keep up with regulated flow recession rates (Figure 9-2). The locations and factors favoring successful cottonwood seedling establishment are discussed below in more detail under Section 9.2.6.

Table 9-1. Generalized timing of Fremont cottonwood life history stages in the Sacramento Valley, California, based on local observations and general knowledge of life history timing in other regions of the western United States. Black shading indicates period of peak activity, medium gray indicates periods of moderate level of activity, and light gray indicates periods of off-peak or low-level activity.

STAGE	Jan		Feb		Mar		Apr		May		June		July		Aug		Sept		Oct		Nov		Dec	
Winter dormancy	■	■	■	■	░	░											░	░	■	■	■	■	■	■
Leaf bud and catkin growth			░	░	■	■	■	░																
Flowering					░	▒	■	■	■	░														
Seed release							░	■	■	■	▒	░												
Germination/initial seedling growth									░	■	■	■	▒	░										
1st summer seedling establishment										░	▒	■	■	■	▒	░								
Tree, sapling and 2nd year seedling growth					■	■	■	■	■	■	■	■	■	■	■	■	▒	▒	░	░				

░	Period of off-peak/low-level activity
▒	Period of moderate activity
■	Period of peak activity

9.2.4 Riparian hydrograph components and cottonwood phenology

In much of western North America, riparian tree species, including Fremont cottonwood, have generally evolved life history strategies that depend on a river's historical hydrology, including the annual cycles of winter floods and spring snow-melt, as well as more infrequent large spring floods during extremely wet years (Figure 9-1).

In order to maintain or reestablish woody riparian vegetation using a process-based restoration approach, managed flows need to mimic natural hydrographs in several key ways

- High flow peaks, which should mimic to some degree the characteristics of peak flows associated with winter peak rain events in the unimpaired hydrograph, are necessary to control vegetation encroachment and prepare seedbeds prior to seedling recruitment flows in wet years (*scouring or encroachment prevention flows* and *seedbed preparation flows*).
- High spring snow-melt peak flows with relatively gradual recession rates during the seed release period for cottonwoods and willows will be needed during wet years to moisten the seedbeds and induce seed germination on geomorphic surfaces suitable for long-term establishment (*recruitment flows* for seedling initiation).
- Summer and fall base flows are needed to ensure that new seedling cohorts and older cohorts of saplings and mature trees have adequate soil moisture for summer growth and survival during the annual dry season (*seedling establishment and maintenance flows*).

The interaction of flow conditions and cottonwood establishment are discussed further in Section 9.2.6, while recommendations for managing flows to promote cottonwood establishment under current regulated conditions are described in Section 9.8.1.1.

9.2.5 Reproduction and establishment

The general life cycle of Fremont cottonwoods, and other tree species, can generally be divided into the following basic stages (Figure 9-3):

- *Initiation*, which includes seed dispersal, germination, and initial seedling growth. In cottonwoods this occurs during the spring and the summer of the first growing season (Braatne et al. 1996, McBain and Trush 2002, Bair 2001, Lytle and Merritt 2004, Stella 2005, Stillwater Sciences 2006).
- *Establishment*, which covers the continued survival and growth of seedlings and saplings for several years until the reach sexual maturity. In Fremont cottonwoods and related species, the first 3 to 4 years appear to be the most critical period for successful establishment (McBain and Trush 2002, Bair 2001, Lytle and Merritt 2004, Stella 2005, Stillwater Sciences 2006).
- *Sexual maturity,* which begins once the tree begins flowering and producing seed. In Fremont cottonwoods this stage typically begins when trees are 5 to 10 years old (Reichenbacher 1984, Braatne et al. 1996, Sawyer and Keeler-Wolf in press).
- *Senescence* begins when the mature tree begins to lose vigor and experiences diminished reproductive output (fewer seeds produced on average per year) and ends with the death of the tree. This may be triggered by mechanical damage, often due to drought stress or storm damage, which may increase susceptibility to various pathogens and other mortality agents. The maximum lifespan of Fremont cottonwood is approximately 130 years in most regions (Braatne et al 1996), although there are some reports of isolated individuals up to 300 years old (Sawyer and Keeler-Wolf in press). Most Fremont cottonwoods die at a

younger age due to fatal diseases or the effects of disturbances such as bank erosion, fire, flood, or drought.

9.2.5.1 Sexual reproduction

Central Valley riparian forest initiation begins with the colonization of bare, moist alluvial surfaces after large floods by seedlings, typically Fremont cottonwoods, willows and other fast-growing species. These pioneer species are physiologically adapted to the highly variable hydrologic and geomorphic regimes of alluvial river floodplain systems. Willows and cottonwoods can sustain high rates of root growth to keep up with rapid ground water decline. Most riparian species are also physiologically adapted to survive prolonged flooding and scour, and they maximize dispersal through high seed output, long seed-floating time, or clonal growth (Johansson et al. 1996, Braatne et al. 1996).

Successful cottonwood recruitment depends on the specific hydrology (flood frequency and duration) of the germination site combined with favorable seed dispersal timing. Site hydrology is a function of river flow, topography, and substrate composition. Seed release timing varies for riparian trees and is often related to their dispersal mechanism; light-seeded, wind-dispersed species tend to release seeds in spring, when newly de-watered banks are exposed, and large-seeded, water-dispersed species tend to release in fall and winter, when seeds can float up onto floodplains (Kondolf et al. 2000, Griggs and Small 2000, McBain and Trush 2002). This combination of hydrologic conditions and seed release timing required for successful establishment of cottonwoods and willows has been formalized by Mahoney and Rood (1998) and others into the 'recruitment box' conceptual model (discussed below in Section 9.2.6).

Under natural conditions, only a fraction of recruited cottonwood seedlings will become established. Annual or seasonal fluctuations in groundwater tables, the timing and magnitude of larger flood events, substrate conditions, and biotic factors (such as competition or herbivory) all influence whether a cohort of seedlings survives long enough to successfully establish a new stand of mature trees. Certain sites, sometimes referred to as "safe sites" or "nurse sites," are more likely than others to provide conditions conducive to successful establishment (Figure 9-2). Field observations and several authors (Bradley and Smith 1986, Braatne et al. 1996, Scott et al. 1996, Kondolf and Wilcock 1996, Friedman et al. 1997, Greco 1999) suggest that floodplain depressions, high flow channels, and other off-channel sites that historically received overbank flooding and sediment deposition provide suitable recruitment conditions as well as protection from later floods under natural conditions. As a result, willow and cottonwood establishment is also episodic, and riparian stand structure is often dominated by several prominent cohorts that established after particular flood events. The coupling of the recruitment box model and the safe site concept provides a useful tool for restoration planning.

Additional conceptual models of cottonwood recruitment, including the "window of opportunity" described by Kondolf and Wilcock (1996) have been proposed by various authors. The window of opportunity model is similar to the recruitment box model in that it relates relative elevation and location of successfully established riparian vegetation to scour and inundation stresses at lower elevations and drought or desiccation stress at higher elevations. The primary difference between the models is that the window of opportunity model focuses on longer-term processes affecting the likely location of mature riparian vegetation and is not focused on particular species, whereas the recruitment box model focuses primarily on the shorter-term processes affecting seed dispersal and germination and first year survival of seedlings of a given species, but also relates elevation of seedlings to sites that are likely to be "safe" from subsequent scour or desiccation, and hence be locations where plants are likely to survive to maturity (see item (b), seedling bank position", in Figure 9-4).

9.2.5.2 Vegetative reproduction

To date, much of the riparian research and process-based restoration efforts on have focused on pioneer species such as various cottonwood species (see Section 9.2.6 and 9.3.1 below). These species, which release their seeds in spring coincident with the historical snowmelt pulse, are most dependent on river hydrology for reproduction and survival and suffer the largest changes in distribution and age structure when the hydrograph is severely altered by flow regulation or major changes in climate (which may be due to natural cycles of climate change or anthropogenic factors causing an elevated rate of global climate change). Although we generally focus on seedling recruitment, vegetative reproduction also occurs in a variety of riparian species, including cottonwoods (Braatne et al. 1996, Tu 2000). High flows occurring at anytime during the year may help to disperse branches or other vegetative fragments to new sites. If these propagules are washed ashore in sites that provide some protection from subsequent high flows, and if suitable soil moisture and receding groundwater levels occur during the root growth period, successful vegetative reproduction may occur. Horticultural restoration techniques for cottonwoods that rely on cuttings take advantage of this trait. Although they may occur at other times, the conditions for vegetative propagule dispersal and successful establishment would most likely occur during wet years, when high winter or spring flows that break off and disperse branches (vegetative propagules) are more likely to occur in association with spring recruitment flows that would allow the roots of newly deposited vegetative propagules to stay in contact with the slowly declining water table.

Although vegetative reproduction is more common in the *Tacamahaca* section (narrow-leaf and black cottonwoods) than in the *Aigeiros* section (Fremont cottonwood and plains cottonwood), under some conditions it may be the dominant form of reproduction in Fremont cottonwood. For example, Tu (2000) found that Fremont cottonwood successfully recruited to the sand splays created by a levee breach on the Cosumnes River primarily via vegetative propagules rather than seed. Along the Sacramento River, however, recruitment of Fremont cottonwood on point bars appears to occur almost entirely via sexual reproduction (i.e. seedling establishment) (Wood 2003a).

9.2.6 The recruitment box model

Mahoney and Rood's (1998) 'Recruitment Box' model "defines the stream stage patterns that enable successful establishment of riparian cottonwood seedlings." Riparian tree recruitment depends on local hydrologic conditions during the seed release period. Early successional species, such as cottonwood and willow, release many seeds that are viable for a short time, typically 2–3 weeks (Braatne et al. 1996) and require bare, moist substrates to germinate. Seedling recruitment, therefore, occurs on the surfaces that happen to be moist and bare during the seed release period. Mahoney and Rood (1993, 1998) describe this window of optimal conditions for riparian plant establishment as the "recruitment box," defined by topographic elevation with respect to river stage and period of seed release and viability (see annotations A and B in Figure 9-4). Within the recruitment box, a further constraint is the maximum survivable rate of water table decline (see annotation C in Figure 9-4). Numerous physiological studies report that seedlings and cuttings of various cottonwood species survive water table declines of 1–1.5 inches/day (2.5–3.8 cm/day). Studies of cottonwood growth have reported that maximum root growth after germination is approximately 1–1.5 inches/day (2.5–3.8 cm/day) (McBride et al.

1989; Mahoney and Rood 1991, 1998; Segelquist et al. 1993). Flow reduction at rates that exceed potential root growth generally results in seedling mortality from desiccation. The recruitment box is constrained at the higher elevations by the seedling's ability to maintain contact with the receding water table following floods, and at lower elevations by inundation and scour the following winter. The model demonstrates why cottonwood establishment is not always successful under natural conditions and how regulated river flow patterns affect seedling establishment. The model is a useful tool to analyze the reproductive ecology of riparian cottonwoods and can allow river managers to prescribe flows to facilitate cottonwood seedling recruitment along dammed rivers. Figure 9-5 illustrates the general conceptual framework for application of the recruitment box model to restoration planning at the site or river corridor scale. This general framework has been used to calibrate and apply the recruitment box model to predict conditions required for successful establishment of Fremont cottonwood, Goodding's willow, and narrowleaf willow along a 150-mile corridor on the San Joaquin River below Friant Dam (Stillwater Sciences 2003), and along the lower Tuolumne River (Stella 2005, Stillwater Sciences 2006).

The key parameters in the recruitment model include maximum root growth rate, height of the capillary fringe and suitable soils, seedling mortality rates, timing of seed dispersal and seed viability, and assumptions about the relationship between river stage and alluvial ground water level. This section summarizes the results of a number of recent studies that collected local data on Fremont cottonwood that can be used to calibrate the recruitment box model for application to the Sacramento River (Roberts et al. 2002, TNC 2003a, Morgan 2005, Morgan and Henderson 2005 a and b). The recruitment flow requirements based on the calibrated recruitment box model have been included in the SacEFT currently under development (ESSA 2005, 2006).

Relative elevation of successful seedling initiation and establishment

The recruitment box model assumes that seedlings in a certain relative elevation zone above the baseflow water surface are likely to experience high mortality from disturbance (scour, inundation, burial under deposited sediments) associated with annual high flow events (i.e., winter storm peak flows). Generally this scour zone elevation band is determined using empirical observations. Several recent studies on the Sacramento River (Roberts et al. 2002, TNC 2003a, Morgan 2005, Morgan and Henderson 2005a and 2005b) have found that cottonwood seedlings often initiate on low surfaces (typically 0–3 ft (0–1 m) above the summer baseflow water surface along the Sacramento) during late spring and summer, but rarely survive to the second summer. Cottonwood seedlings that survive to become successfully established trees typically initiate on slightly higher relative elevation surfaces. Recent work at three point bars between RM 192-172 indicates that successful cottonwood recruitment occurs at relative elevations of 3–9 ft (1–3 m) above summer baseflow levels (Roberts et al. 2002, TNC 2003a). Similar results have been observed along the lower Tuolumne and San Joaquin rivers, although successful recruitment band in these smaller rivers tends to be at slightly lower elevations of 2–6 ft (0.6–2 m) (McBain and Trush 2002, Stillwater Sciences 2003 and 2006, Stella 2005).

Seed dispersal timing

Observations of seed release timing of Fremont cottonwood, arroyo willow (*Salix lasiolepis*), and Gooddings's willow (*S. gooddingii*), were conducted in 2000 along the Sacramento River by Peterson (2002) (Figure 9-6). Morgan (2005) observed seed release timing of Fremont cottonwood along two point bars (RM 192 and 183) in the Chico Landing to Colusa sub-reach during 2003 and 2004. Similar observations of Fremont cottonwood, Goodding's willow and narrowleaf willow (*Salix exigua*) were made along the Tuolumne and San Joaquin rivers during 2002–2004 as part of a study to calibrate the recruitment box model for application to the San

Joaquin Basin (Stillwater Sciences 2003 and 2006, Stella 2005, Stella et al. in press)(Figure 9-7). Other more qualitative estimates of seed release timing for cottonwoods and other species for the Central Valley are provided by Griggs and Small (2000), Kondolf et al. (2000), and McBain and Trush (2002).

Based on observations made in the studies cited above, the following windows of seed dispersal timing are presented for use in application of the recruitment box model to the Sacramento River:

Arroyo willow	March–April (peak period: March 15–April 15)
Fremont cottonwood	April 15–July (peak period: last week of April–first week of June))
Goodding's willow	May 15–August (peak period: June 1–July 15)
Narrowleaf willow	June–August (peak period: June 1–July 15)

The multi-year observations in the San Joaquin Basin (Stella 2005, Stillwater Sciences 2006, Stella et al. in press) indicate some variability in seed release timing from year-to-year (Figure 9-7), presumably caused by annual variations in weather. The San Joaquin Basin studies also indicate much site-to-site variability in seed release timing. The periods indicated above, should therefore be treated as general patterns to be expected along the Sacramento River. Collection of site-specific data should improve accuracy of timing estimates for particular sites along the river that might be targeted for restoration via natural recruitment processes.

Degree-day modeling, which accounts for cumulative heat load above some minimum temperature threshold, offers a means of improving predictions of peak seed release timing during any particular year by accounting for local weather condition, at least for Fremont cottonwood and Goodding's willow in the San Joaquin Basin (Stella 2005, Stillwater Sciences 2006, Stella et al. in press). Morgan (2005) also suggests that degree-day modeling can reduce variability in predictions of seed release timing for Fremont cottonwood along the Sacramento River.

Seedling mortality related to stage recession rates

Numerous studies have reported adequate cottonwood and willow survival at water table decline rates of 1 to 1.5 in/day (2.5 to 3.75 cm/d), however many of these studies used stem cuttings rather than recently germinated seedlings or did not measure root growth rate directly, but inferred them from water table manipulation rates and survival data (McBride et al. 1989, Segelquist et al. 1993, Mahoney and Rood 1992 and 1998, Amlin and Rood 2002). In contrast, experiments conducted using young Fremont cottonwood and Goodding's and narrowleaf willow (*Salix gooddingii* and *S. exigua*) seedlings grown from seed collected in the San Joaquin Basin (along the Tuolumne and San Joaquin rivers) by Stella and colleagues in summer 2002 indicate that actual average root growth rates can be substantially less for young seedlings (Stella 2005, Stillwater Sciences 2006, Stella et al. in review). Morgan (2005) tracked seedling cohorts on two point bars on the middle Sacramento River and estimated cottonwood seedling root growth rates during the first summer. She found average root growth rates of 0.2 in/day (5 mm/day), with minimum rates of 0.1 in/day (2 mm/day) and maximum of 0.6 in/day (14 mm/day), when root growth rates were averaged over the full summer growing season. Roberts and other (2002), working on the same point bars, tracked seedling root length throughout the summer. They found 7 week old seedlings had average root lengths of about 16 in (40 cm), with maximum measured root lengths of almost 24 in (60 cm). Their data also indicate that maximum root growth rates generally occur early in the summer when they found average root growth rates of 0.9 in (23 mm/day) and maximum rates of 1.3 in (32 mm/day). Root growth rates appeared to decrease later in the summer, but root lengths of older seedlings may have been underestimated because of difficulties in excavating beyond 1.6 ft (0.5 m) under field conditions (Roberts et al. 2002). In spite of this potential for underestimating root growth rates, they found average growth rates of around 0.3 in/day (8 mm/day), and maximum growth rates of 0.5 in (12 mm/day), over a 7-week period in the summer. These results are similar to those subsequently obtained by Morgan (2005).

The experimental results of Stella et al. (in review) indicated that Fremont cottonwood seedlings were less tolerant of water stress caused by rapid water table decline than were the two willow species (Figure 9-8). Of the three species tested, Goodding's willow was the most tolerant and showed higher overall survival (Figure 9-8), root elongation (Figure 9-9), shoot growth rates, and biomass accumulation (Stella et al. in review). These experimental results, and the recent field studies on Sacramento River point bars (Morgan 2005, Morgan and Henderson 2005a and b) indicate that successful establishment of large cohorts of Fremont cottonwood seedlings is most likely to occur when water table/river stage declines at average rates of less than 0.8 in/day (2 cm/day) (Stella et al. in review, Stillwater Sciences 2006, Stella 2005, Morgan 2005, Morgan and Henderson 2005b). These same studies indicate that rates of decline in the range of 0.8 to 1.6 in/day (2-4 cm/day) are stressful to seedlings, but may still support survival of a smaller cohort of seedlings. Rates of water table/river stage decline greater than 1.6 in/day (4 cm/day) are highly stressful and are likely to lead to complete loss (100% mortality) of first year seedlings. Morgan and Henderson (2005b) suggest that some day-to-day variation in stage declines may be acceptable, and that 3-day running averages should be used in establishing appropriate rates of stage decline as suggested by Rood and others (2003, 2005). It is also possible that steeper rates of river stage recession may be acceptable if they are offset by periods of 1 or more days of stable water levels, which would produce a stepped recession limb of the recruitment flow hydrograph (TNC 2003a, Stillwater Sciences 2006).

The results of Stella and colleagues (Stella et al. in review, Stillwater Sciences 2006, Stella 2005) further suggest that rates of decline in the range of 0.8 to 1.6 in/day (2–4 cm/day) are likely to result in higher percent survival of willow seedlings, particularly Goodding's willow, than cottonwood seedlings. For example, Stella et al. (in review) found that under recession rates of 1.2 in/day (3 cm/day), survival of seedlings to day 50 was greatest (35%) for Goodding's willow, intermediate (26%) for narrowleaf willow, and lowest (12%) for Fremont cottonwood. Thus, some recruitment flows that might be relatively unsuccessful for cottonwood recruitment because of moderately rapid stage recession rates that are too stressful for cottonwood seedlings might still result in moderate levels of willow recruitment. This may explain the observations of willow recruitment being more common than cottonwood recruitment under current conditions on point bars of the middle Sacramento River (TNC 2003a, Wood 2003a).

Effects of soil/substrate type

Soil texture and chemical characteristics can substantially affect seedling germination success, growth rates, and mortality rates. Soil texture has important effects on the availability of water to plant roots. Of particular importance is the influence of soil texture on the height of the capillary fringe zone above the alluvial ground water table. Capillary fringe height generally increases as soil texture decreases.

Based on observations at various sites along the Tuolumne and Merced rivers (Stella 2005, Stillwater Sciences 2006) and along the Sacramento River (Morgan 2005), it appears that a conservative estimate of the average depth of the capillary fringe zone would be 0.8–1 ft (0.2–0.3 m). This likely underestimates capillary fringe depth in many if not most suitable recruitment sites, but local field measurements of soil texture, ground water, and capillary fringe would be needed to validate or modify this parameter value for application of the recruitment box model to a particular reach or site.

Observations of successful recruitment sites along the middle Sacramento River indicate that cottonwood and willow seedlings are more likely to become established in sites with finer substrates, primarily silts and clays, which would support increased depths of capillary fringe zones. Such sites are more likely to occur on the downstream end of point bars (TNC 2003a, Wood 2003a, Morgan 2005) and in oxbow lakes (Greco 1999, Fremier 2003, Wood 2003b) as they become “terrestrialized” (see in Figure 8-2 and Chapter 8 for a description of the terrestrialization process). Morgan (2005) found that favorable recruitment microsites with finer substrates on point bars typically had capillary fringe zone depths of approximately of 2 ft (0.6 m), compared to the typical 0.8–1 ft (0.2–0.3 m) depths on less suitable areas on the same point bars.

Soil salinity is another factor that can exert a strong influence on vegetation recruitment and establishment. Most native woody species occurring in Central Valley riparian zones have little tolerance for saline soils (Stillwater Sciences 2003). Stillwater Sciences (2003) analyzed the relationship between soil salinity and current vegetation along the San Joaquin River and concluded that recruitment and establishment of cottonwood, willows, and other woody species found in local riparian forests generally occur only on soils mapped as having zero (“free”) or low salinity (Stillwater Sciences 2003). However, under current conditions along the middle Sacramento River it seems unlikely that soil salinity would limit cottonwood recruitment.

Other assumptions required for model application

In applying the recruitment box model it is generally assumed that the elevation of the shallow alluvial ground water table is equal to the surface water elevation in the river. In the absence of detailed site information, and for the purposes of corridor-scale restoration planning, this assumption seems reasonable. Recent data collected at two point bars as part of the DWR studies (Morgan and Henderson 2005a and 2005b), and the earlier TNC studies at the same sites (Roberts et al. 2002, TNC 2003a), indicate that this assumption is generally valid for potential point bar recruitment zones located near the river channel. However, at sites further away from the main channel, groundwater elevations may deviate from water surface elevation in the main channel. In the case of the two study sites, groundwater elevation increased away from the channel indicating the sites are located in a “gaining” reach (i.e., there is a net inflow of groundwater into the channel). In addition, at the two study sites, groundwater level fluctuations tend to match fluctuations in river stage at sites closest to the channel, and are more muted at sites farther from the channel.

Another assumption is that suitable seedbed areas will be available in appropriate spots when recruitment flows occur. If peak flows of sufficient magnitude occur prior to a recruitment flow, sites within the recruitment band will likely be free of vegetation and provide suitable seedbed conditions (bare mineral sediments) due to scour and fine sediment deposition associated with the peak flows. If opportunity for a managed recruitment flow occurs in a year that did not have high winter peak flows, active (mechanical) clearing of targeted recruitment sites might be required to promote desired levels of cottonwood seedling initiation and establishment, although supplemental weed control may also be required if the site currently supports non-native vegetation with a well-developed seed bank (Stillwater Sciences 2003 and 2006, Peterson 2002, Jamieson and Braatne 2001). Significant appropriate seedbed appears to be currently available along the middle Sacramento River, suggesting that seedbed creation via winter and spring peak flows is not likely to be a critical factor limiting cottonwood recruitment under current conditions (Roberts et al. 2002, TNC 2003, Wood 2003a, Morgan 2005, Morgan and Henderson 2005ab).

9.3 Riparian Vegetation Dynamics

9.3.1 Riparian zone plant succession and structure

At a fine scale, riparian zones can be seen as non-equilibrium ecosystems, in which patches of vegetation become established and are seasonally altered (and often extinguished) by inputs of water and nutrients and by deposition and scour of sediment (Bendix 1994, 1999; McBride and Strahan 1984; Stromberg et al. 1991; Stromberg 1997). At a coarser scale, riparian corridors can be seen as a steady-state landscape, in which the formation and annihilation of vegetation patches balance out over the long-term, resulting in a shifting mosaic (or spatially heterogeneous, temporally dynamic patchwork) of habitats that have evolved under the influence of frequent disturbance (Johnson et al. 1976, Wiens 2002). The natural mosaic of a mature riparian zone along the Sacramento River, for example, consisted of willow scrub and cottonwood forests on lower elevation sites, with mixed riparian forests occupying mid-elevation floodplain sites, and valley oak woodland and savannah occupying the oldest and driest floodplain sites (i.e., high terraces frequently located above cut banks) (Vaghti and Greco in press, Greco et al. in press, Vaghti 2003, Greco 1999, Katibah 1984).

Riparian forests require periodic seedling recruitment and subsequent establishment to replace mature and dying trees, maintain the stand through time, and reset the process of vegetation succession. Recruitment (also known as initiation) refers to seedling germination following seed release. Establishment refers to the life stage when a plant has developed sufficient root-and-shoot architecture to survive annual environmental conditions (especially inundation, scour, and drought) and develop into a reproducing adult. Succession refers to a progressive replacement of different plant communities over time in response to internal competition among different plant species or outside disturbances such as floods and fire (Oliver and Larson 1996, Malanson 1993). In classical succession, an orderly, deterministic and directional progression of plant communities is assumed to occur in the absence of major disturbance. In many natural ecosystems, changes in vegetation types over time appear to be much more stochastic or unpredictable, and may be determined largely by the pool of available colonizers (viable seeds or vegetative propagules) present immediately after an initial resetting disturbance event, with founder effects driving the structure and composition of the vegetation that develops and persists on the site more than local environmental factors (Malanson and Butler 1991). Some authors have suggested that riparian communities should be treated as compositionally stable plant assemblages that are maintained by periodic flooding, rather than successional communities that are in some stage of recovery from flood disturbance (e.g., Sigafoos 1964, Hupp 1983, Bendix 1998, Bendix and Hupp 2000).

Riparian vegetation dynamics are tightly coupled with riverine processes; flooding, scour, and sediment deposition strongly influence riparian plant species composition, distribution, and physical structure and are major drivers of riparian community succession. Along geomorphically active, meandering streams, riparian vegetation typically exhibits two distinct patterns or pathways of initial establishment: colonization on bare substrates associated with point bars created by progressive channel migration versus similar bare substrates associated with abandoned channels or oxbow lakes and sloughs created by meander cutoffs and channel avulsion. Cottonwoods and willows are typically among the first species to colonize bare stream banks and bars. These species have high seed output and rapid growth rates—physiological traits that are well suited for quickly colonizing freshly deposited or scoured material. Cottonwoods and willows tend to establish in bands parallel to the channel, with age increasing with distance away from the active channel margin (Figure 9-10) (Gregory et al. 1991, McBride and Strahan 1984, Walker and Chapin 1986). Each band of vegetation represents a separate recruitment event; the position and shape of the stand reflect the favorable flow and sediment conditions (usually a spring flood recession event) that occurred during the spring seed release period. As vegetation from one cohort matures, it traps sediment and extends the bar surface, creating new seedbeds for successive recruitment events (Johnson et al. 1976, Strahan 1984, Scott et al. 1996). In addition to the establishment of pioneer species on newly deposited floodplain and bar surfaces, subsequent hydrologic, geomorphic, and ecological or successional processes alter vegetation composition in established riparian stands in a fairly predictable manner (Figure 9-11). Over time, pioneer vegetation traps sediment and adds litter and nutrient inputs to floodplain soils (Walker and Chapin 1986). As the floodplain develops and the riparian stand ages, other riparian species such as Oregon ash (*Fraxinus latifolia*), box elder (*Acer negundo*), and valley oak (*Quercus lobata*) establish within the riparian zone. These "later successional" species typically produce larger seeds and are more shade-tolerant than the early pioneers, which allows them to persist in the seedbank and germinate under the forest canopy when soil temperature and moisture conditions are adequate (Table 9-2). Recruitment of these species is not as dependent on flow and sediment conditions as for the willows and cottonwoods, and seedling recruitment typically occurs as chance events, depending on individual conditions such as microclimate and proximity to parent trees. Over time, these species further alter the soil, light, moisture, and nutrient conditions within the riparian zone and outlive or outcompete the original pioneer species. At any one site, the spatial and temporal patterns of physical processes (such as flooding and sediment dynamics) and biological processes (such as plant establishment and competition) can be complicated and unpredictable, and vegetation composition is often more patchy than the generalized patterns described above. Recent studies by Greco (1999), Trowbridge (2002), Wood (2003b) and Fremier (2003) indicate that the basic facilitation model of succession, as described above, which has been frequently proposed for Central Valley riparian systems may be too simplified and deterministic. Actual vegetation dynamics may be much more complicated, with multiple types of middle and later seral stages possible with outcomes governed largely by local site conditions (e.g., soil texture, soil stratification, and depth the groundwater), the physical pathway that creates new surfaces (e.g., gradual meander migration versus cutoff events or channel avulsion), or by historical factors (e.g., the seasonal timing of the initial resetting disturbance event, the pool of seeds and vegetative propagules available immediately after the disturbance event, founder effects, potential influence of non-native species). TNC's (2003b) analysis of factors affecting revegetation success at restoration sites provides further evidence of the importance of fine-scale, site-specific factors affecting vegetation development, particularly soil texture, soil profile stratification, and depth to groundwater (Figure 9-12). It is quite possible that multiple stable states may occur rather than a single climax plant community (Baker and Walford 1995).

The above discussion focuses primarily on one successional pathway: recruitment and establishment of pioneer species, primarily cottonwoods and willows, on newly created bar and floodplain surfaces, setting in motion a successional process that can lead to diverse and structurally complex mixed riparian forests. Examination of historical maps and aerial photographs to generate a time-series analysis of riparian vegetation patterns often suggests that in many Central Valley alluvial systems, there is an alternative pathway: channel avulsion or meander cut-offs may create abandoned channels and oxbow lakes. Under certain conditions, these abandoned channels or infilling oxbow lakes develop areas of relatively exposed mineral substrates and saturated surface soil conditions during spring and summer that make them suitable for germination and establishment of pioneer species such as willow and cottonwood. These patches may then under subsequent succession and structural development to create mixed riparian forest. Greco (1999), for example, has documented the importance of this pathway, along with the point bar recruitment pathway, in a reach of the Sacramento River near Chico over a 60-year period. Deeper oxbow lakes tend to persist for longer periods than shallower lakes and may support a mixture of valuable freshwater marsh, open water, and seasonal wetland habitats.

Table 9-2. Riparian plant species ecological needs. Source: adapted from Griggs and Small 2000.

Species	Location on floodplain	Light needs	Water table needs	Drought tolerance	Seed dispersal
Fremont cottonwood	Point bars and Cutoff Channels	Full Sun; poor growth in shade	Must have roots in top of water table	None	Spring
Valley, arroyo, yellow, sandbar willows	Point bars and Cutoff channels	Full Sun; poor growth in shade	Must have roots in top of water table	None	Spring-Summer
Oregon ash and box-elder	Usually away from active channel on low to medium floodplain surfaces	Tolerates shade	Facultative	Drought tolerant in shade	Fall
California sycamore	Along secondary channels and ox-bow lakes	Full sun; tolerates some shade	Must have roots in top of water table	Resprouts from crown	Winter
White alder	Ox-bow lakes	Full sun	Must have roots in top of water table	None	Fall
Buttonbush	Ox-bow lakes	Tolerates shade	Must have roots in top of water table	Resprouts from crown	Fall
Valley oak, elderberry, rose	High floodplain surfaces	Tolerates shade	Facultative	Well-developed	Fall
Arundo (non-native)	Point bars, floodplain surfaces	Tolerates shade	Facultative	Well-developed	Winter-Spring (vegetative dispersal)

Table 9-3 provides a conceptual model of the effect of hydrology and fluvial geomorphology on woody vegetation in lowland, alluvial rivers in the Central Valley, including the Sacramento River and its major tributaries.

Table 9-3. Summary matrix of interactive effects of individual hydrograph components and fluvial geomorphology on woody riparian vegetation in lowland alluvial rivers of the Central Valley. Source: Adapted from McBain & Trush (2002).

Geomorphic feature ⇨ **Hydrograph component ⇩**	**Point bar**	**Floodplains**	**Terraces**	**Outside of meander bends**	**Oxbows**
Winter/ spring baseflow	Promote inundation mortality of seedlings. Prevent germination by inundating the active channel margins.	Maintains or recharges ground water, promoting late season growth and maximum growth after plants break dormancy.	Maintains or recharges shallow groundwater aquifer, facilitating maximum growth in establishing, mature, and senescent vegetation.	Maintains or recharges shallow groundwater aquifer.	Maintains or recharges shallow groundwater aquifer, sustaining off channel wetlands.
Winter floods	Significantly mobilize channel bed, scouring seedlings initiated in the previous year. Disperses vegetative propagules.	Builds and adds nutrients to floodplain by fine sediment and organics deposition. Promotes inundation mortality in physiologically sensitive plant species. Deposits seeds, establishes short term seed bank waiting for suitable germination conditions. Disperses vegetative propagules.	Deposits seeds, establishing a short-term seed bank for future suitable germination conditions. Disperses vegetative propagules.	Channel migrates against the outside of the bend, causing limited mortality to mature and senescent vegetation, introducing large woody debris.	Overbank flow can refill sloughs and oxbows, potentially introducing additional plant species.
Extreme winter floods (during normal or above normal water years)	Move and reorganize in-channel woody debris. Realign channel by avulsion or cutting off sharp meander bends creating abandoned channels, sloughs, and oxbow lakes. Disperses vegetative propagules.	Scour or topple mature and senescent vegetation, creates new seed beds. Mobilize wood jams. Fine sediment deposition promotes root suffocation of certain species. Disperses vegetative propagules.	Builds and add nutrients to terrace by fine sediment and organics deposition. Promotes inundation mortality in physiologically sensitive plant species. Fine sediment deposition promotes root suffocation	Channel migrates against the outside of the bend, causing limited mortality to mature and senescent vegetation, introducing large woody debris.	Oxbow may be recaptured by the channel and the wetland reoccupied by the main channel. Fine sediment and organics deposition creates greater topographic variation and increases nutrient availability.

Geomorphic feature ⇨ Hydrograph component ⇩	Point bar	Floodplains	Terraces	Outside of meander bends	Oxbows
			of certain species.		
Snowmelt peak	Prevent germination by inundation of point bar. Scour establishing seedlings. Promote inundation related mortality (drowning).	Encourages seed germination by providing high soil moisture. Discourages germination near the active channel by inundation.	Encourages seed germination by providing high soil moisture.	Channel migrates against the outside of the bend, causing limited mortality to mature and senescent vegetation, introducing large woody debris.	Surface and groundwater recharges, creating the specialized environmental conditions required by ephemeral herbaceous plant species.
Snowmelt recession	Prevent plant germination by inundation. Promote germination by creating moist seedbed conditions as point bar surfaces become exposed.	Facilitates seed germination over a wide elevation range. Drops in river stage causes desiccation mortality to plants that had germinated earlier in the spring.	Drops in river stage causes desiccation mortality to plants that had germinated earlier in the spring.	Recharges ground water promoting maximum growth after breaking dormancy.	Surface and groundwater recharges, creating the specialized environmental conditions required by ephemeral herbaceous plant species. Water table draw down causes desiccation related mortality.
Summer baseflows	Facilitates plant germination on point bar late in the growing season, encouraging scour or inundation mortality during high flows in the subsequent winter and spring. Sustains herbaceous perennials along the summer baseflow water surface elevation. Sustains shallow alluvial groundwater table near the channel (especially in losing reaches).	Desiccate seedlings that germinated during the late winter and spring on higher surfaces. Helps maintain shallow alluvial groundwater table.	Low water tables stresses plants, leading to desiccation related mortality.	Desiccate seedlings that germinated through the late winter and spring.	In below normal water years, some portions could dry up, causing widespread mortality to aquatic and emergent vegetation.

9.3.1.1 The fundamental role of pioneer riparian species

Changes in sediment dynamics and the magnitude, frequency, and timing of river flows have the largest and most immediate impact on pioneer riparian communities. In the Sacramento River system, the dominant woody pioneer species affected are Fremont cottonwood, Goodding's black willow, narrowleaf willow (also known locally as sandbar willow), shining willow (*S. lucida*), and arroyo willow (*S. lasiolepis*) (Vaghti 2003, Vaghti and Greco, in press). Understanding the distribution and reproductive ecology of these species is important to river management for several reasons. All are pioneer species in the family Salicaceae, though the species exhibit differences in reproductive timing and method (seed versus vegetative), habit (tree versus shrub), and topographic distribution (floodplain versus point bar). Contemporary management and restoration objectives most often seek to balance the desire to increase floodplain riparian forest area to include a diversity of native woody species, with the need to maintain a dynamic, meandering riverbed by preventing encroachment of pioneer vegetation (primarily narrow-leaf willow) into the active channel.

Central Valley pioneer riparian tree species have evolved reproduction strategies that depend on the timing and magnitude of river flow conditions during the historical spring snowmelt period. Cottonwoods and willows release many seeds in spring that are viable for a short time, typically a few weeks, and require bare, moist substrates to germinate. On sloping point bars and river banks, recruitment events often occur in narrow bands that are constrained at the higher elevations by the seedlings' ability to maintain contact with the receding water table following spring floods, and at lower elevations by prolonged inundation and frequent scour during subsequent high-flow periods. River regulation changes flow timing, magnitude, and rate of recession and has the potential to reduce the viability of riparian seedlings. Availability of water to developing seedlings is particularly critical during summer in the Central Valley because high temperature and relatively low humidity in the typically very exposed seedling recruitment sites combine to create conditions leading to the potential for very high plant water loss rates, which means that seedlings that become decoupled from ground water sources experience rapid desiccation and subsequent mortality.

Previous work conducted on the Sacramento, Tuolumne, Merced, and San Joaquin rivers documents that cottonwood and willow seedlings germinate readily on low bars in these rivers, but rarely survive to reproductive maturity (Morgan 2005, Morgan and Henderson 2005b, Roberts et al. 2002, TNC 2003a, McBain and Trush 2000, Stillwater Sciences 2001, Stella et al. 2003, Stillwater Sciences 2003 and 2006, Stella et al. in press). There are very few sapling cohorts in evidence along these rivers, suggesting that recruitment and subsequent survival of these species, particularly Fremont cottonwood, are impaired. Changes in flow magnitude and timing due to river regulation have important consequences for riparian species, especially pioneers, whose life cycles are typically timed with annual spring runoff peaks.

9.3.2 Fremont cottonwood as a foundation species

Some of our current understanding of riparian habitat dynamics in the middle Sacramento River corridor and hypotheses regarding the effects of land and water management on cottonwoods and other riparian species are guided by a conceptual model, adapted and modified by Battles et al (2005) from Strange et al. (1999), of riparian community development (Figure 9-13). In the model ecosystem components are classified as drivers, processes, patterns, and ecosystem functions. Climatic factors (i.e., precipitation and temperature) and basin characteristics (e.g., latitude, area, elevation, topography, and parent material) are the ecosystem drivers, and are analogous to state factors in other ecosystem models (Jenny 1941, Likens et al. 1970, Groffman et al. 2004).

Our study concerns the ecological processes and patterns (the shaded box in Figure 9-13) that result from the interaction of these drivers over annual and decadal scales. The most influential of these processes is flow regime, specifically flow timing, magnitude, and sediment dynamics. These processes determine the potential distribution (e.g., geographic range and population age structure) of cottonwood for the Central Valley. Actual distributions are narrowed further by biotic interactions and human modification of the landscape and flow regime. Biotic interactions such as competition and herbivory are generally considered less important in structuring this non-equilibrium, disturbance-driven ecosystem than physical factors and dispersal (Stillwater Sciences, unpublished data; Mahoney and Rood 1998; Johnson 2000).

Cottonwoods dominate the early-successional phase of riparian community development. Therefore the ecological properties of these populations (i.e., size structure, age distribution, density, and growth rate) serve as the landscape template on which the riparian ecosystem develops. This interaction between process and pattern governs important riparian functions such as energy inputs, habitat structure, microclimate modification to the instream and riparian environments, large woody debris production, and streambank stabilization.

Major human impacts to the ecosystem occur at all levels of the model by modifying drivers, processes, community structure, and landscape patterns. These are indicated as external inputs in the conceptual model (Figure 9-13). The most important alterations are to the climate (via global warming and consequent changes in precipitation and temperature influencing the natural hydrologic regime), anthropogenic flow regulation (with consequent changes in river hydrology and sediment regime), and landscape modification, such as agricultural conversion, levee construction and bank protection.

9.4 Riparian-Associated Songbirds

Fremont cottonwood forests, and other riparian vegetation types, provide habitat for a wide variety of wildlife species, including a number of riparian songbirds. This section provides a brief overview of our current understanding of riparian habitat-songbird relationships in the Sacramento Basin.

9.4.1 Wildlife species found in riparian habitats

The loss of riparian forest has reduced habitat for terrestrial wildlife species and birds, thus diminishing their general abundance and distribution in the river corridor. In California, over 225 species of birds, mammals, reptiles, and amphibians depend on riparian habitats, and riparian ecosystems harbor the most diverse bird communities in the arid and semi-arid regions of the western United States (Knopf et al. 1988, Dobkin 1994, Saab et al. 1995). In addition to high species richness, riparian areas during the bird breeding season can harbor individuals at densities up to ten times greater than the surrounding terrestrial habitats (RHJV 2000, 2004).

A variety of invertebrates occur in riparian habitats along the Sacramento River. One of these is the valley elderberry longhorn beetle (*Desmocerus californicus dimorphus*), which is federally listed as threatened and has therefore received a fair amount of attention in the Central Valley (Huxell 2000, Collinge et al. 2001, Morrison et al 2003, River Partners 2004). Elderberries (*Sambucus mexicanus*) are the sole host plant for oviposition and larval rearing. Hunt (2004) studies groundbeetle assemblages in remnant (> 30years old) and restored age (1-3 year old and 6-10 year old restoration sites) riparian forests along the middle Sacramento River and found a strong response of assemblages to forest age. Terrestrial and aquatic invertebrates are also important in food web interactions and nutrient subsidies between aquatic and riparian ecosystems (Power and Rainey 2000), which can be strongly influenced by flood regimes (Bonn et al. 2002).

Other recent studies have focused on various vertebrate species (such fish, amphibians, reptiles, rodents, bats, birds) associated with riparian and floodplain habitats along the Sacramento River. Golet and colleagues (2003) and TNC (2005) summarize a number of these recent studies. The discussion below (Section 9.4.2) focuses on birds, particularly songbirds, that currently or historically where known to use riparian habitats along the Sacramento River.

9.4.2 Habitat criteria for riparian songbirds

Conservation and restoration of riparian and wetland habitat is of primary importance for supporting populations of wildlife and fish associated with the river-riparian corridor. Although quantitative information on wildlife habitat requirements (e.g., minimum patch size, distance to foraging habitat, distance to water) is largely unavailable, we summarized quantitative criteria specifically for yellow-billed cuckoo (Section 9.4.2.1) based on literature reviews and consultation with experts, and provide a brief overview of current understanding of habitat relationships for a variety of other riparian songbirds based on recent and ongoing research in Central Valley riparian ecosystems (Section 9.4.2.2).

Based on available information, habitat criteria have been best quantified and modeled for yellow-billed cuckoo, a willow-cottonwood riparian forest obligate species. Yellow-billed cuckoo is thus often selected as a riparian focal species because of its special status and because existing habitat-relationship information allows patch size analysis to be conducted for this species. While the habitat characteristics presented below are specific to yellow-billed cuckoo, we believe that a number of other riparian forest species would benefit if habitat for yellow-billed cuckoo is enhanced along the middle Sacramento River. Section 9.4.2.1 summarizes some of the known and hypothesized habitat relationships for the yellow-billed cuckoo.

However, the recent scientific literature provides many reasons why use of a single indicator, umbrella or surrogate species may lead to erroneous conclusions and potentially misguided restoration and management actions (Landres et al. 1988, Simberloff 1998, Andelman and Fagan 2000). The Riparian Habitat Joint Venture (2004) discusses this issue in more detail and provides good justification for the use of a suite of riparian songbirds as focal species to help guide assessment and management of riparian ecosystems in California. In recognition of the limitations of relying on a single species to represent the habitat needs of riparian-associated songbirds, Section 9.4.2.2 summarizes recent and ongoing studies being conducted by PRBO Conservation Science on bird-habitat relationships in Central Valley riparian systems (e.g., Gardali et al. 2005 and 2006, Nur et al. 2005), which are leading to an increased understanding of species- and region-specific responses to riparian habitat conditions. Ultimately, such understanding is needed to insure that long-term management actions can be selected that will promote maintenance of viable populations of the full array of riparian songbirds (and other wildlife species) that currently or historically occurred in the middle Sacramento River corridor.

9.4.2.1 Yellow-billed cuckoo

Cuckoos typically inhabit densely foliated, deciduous trees and shrubs, particularly willows, with a dense understory formed by blackberry, nettles, and/or wild grapes, adjacent to slow-moving watercourses, backwaters, or seeps (CDFG 1983). River bottoms and other mesic habitats, including valley-foothill and desert riparian habitats, are necessary for breeding. Dense low-level or understory foliage with high humidity is preferred (Gaines 1974, 1977).

Field studies and habitat suitability modeling have concluded that vegetation type (e.g., cottonwood-willow forest), patch size, patch width, and distance to water are important factors determining the suitability of habitat for yellow-billed cuckoo breeding (Table 9-4) (Laymon and Halterman 1989, Greco 1999). Patch size was the most important variable determining presence of cuckoos on the Sacramento River from 1987 to 1990 (Halterman 1991, as cited in Laymon 1998), with a trend toward increasing occupancy with increased patch size. Patch sizes greater than 198 acres (80 hectares) were always occupied by cuckoos, with 59% occupancy in patches 101 to 198 ac (41 to 80 ha) in extent, and only 10% occupancy in patches 50 to 100 ac (20 to 40 ha) in extent (Laymon and Halterman 1989). Few cuckoos have been found in forested habitat of less than 25 ac (10 ha) (Anderson et al. 1994). Willow-cottonwood habitat patches greater than 1,970 ft (600 m) in width were found to be optimal, and typically anything less than 328 ft (100 m) was unsuitable (Laymon and Halterman 1989). Halterman (1991, as cited in Greco 1999) and Laymon et al. (1997, as cited in Greco 1999) also observed nesting more frequently in areas where the distance to water was less than 328 ft (100 m). Dense vegetation less than 66 ft (20 m) in height is especially important for nesting, while lower and higher vegetation with greater overall foliage density is used for foraging (Laymon et al. 1997, as cited in Greco 1999).

Table 9-4. Habitat suitability of yellow-billed cuckoos in California.

Habitat suitability	Habitat type	Area		Width		Patch distance to water	
		ac	ha	ft	m	ft	m
Optimal	Willow-Cottonwood	> 198	> 80	> 1,969	> 600	< 328	< 100
Suitable	Willow-Cottonwood	101–198	41–80	656–1,969	200–600	-	-
Marginal	Willow-Cottonwood	42–99	17–40	328–653	100–199	-	-
Unsuitable	Willow-Cottonwood	< 42	< 17	< 328	< 100	> 328	> 100

Source: Greco 1999 (adapted and modified from Laymon and Halterman 1989; Laymon et al. 1997)

Young, rapidly growing stands of riparian vegetation provide preferred nest sites and high productivity of invertebrate prey, with a lower prevalence of predators compared with older stands (Laymon 1998; Halterman 1991, as cited in Laymon 1998). Greco (1999) defined this to be less than 45–60 years since vegetation became established on newly formed substrate, stressing the importance of meandering riparian systems with intact erosional and depositional processes that create new areas for riparian vegetation to establish.

Other important habitat characteristics include canopy height, canopy cover, and understory cover (see Suckling et al. 1998 for summary). Halterman (1991, as cited by Laymon 1998) found that habitat fragmentation, as determined by the extent of habitat per 5-mi (8-km) river reach, was the second most important variable (after patch size) in determining the presence of cuckoos, followed by the presence of low woody vegetation. Other evidence indicates that a general focus only on cottonwood-willow patch area and width may be insufficient, and that proximity to water and the presence of a mosaic of other habitats may be important factors affecting habitat use by cuckoos along the middle Sacramento River (J. Silveira, pers. comm. 2005).

The minimum goal for a viable subpopulation has been cited in the literature as 25 pairs, with interchange to other subpopulations (Laymon and Halterman 1989, Laymon 1998). This goal was adopted by The Nature Conservancy and The National Audubon Society in the habitat management plan for the cuckoo at the Kern River Preserve (Laymon and Halterman 1989).

Because riparian systems in California are extremely dynamic—due to natural channel migration and floodplain dynamics, the rapid growth rates of riparian vegetation, and human land use practices—the habitat where cuckoos live can change rapidly over time (Greco 1999). Conservation and restoration efforts need to keep in mind that large areas need to be conserved to allow for the natural formation and loss of habitat. Management strategies involving "minimum dynamic areas" (Pickett and Thompson 1978, as cited in Greco 1999), such as those discussed in the Sacramento River Conservation Area Handbook (SRCAF 2003), are preferred over conservation of minimum patch size areas (Greco 1999). Restoration should be geared toward maintenance of channel hydrodynamic processes that result in formation of complex riparian habitat (Greco 1999).

Most extant riparian habitat is located in the primary floodplain, which suggests that large flood events that inundate primary foraging habitat could be detrimental to cuckoos, especially if such events occur during the nesting season (Laymon 1998). Restoration efforts that develop a portion of riparian forests on upper floodplain surfaces and terraces should benefit cuckoos by providing potential refugia and suitable foraging habitat during large flood events.

9.4.2.2 Other riparian songbirds

The RHJV (2004) selected a suite of birds to serve as focal species for assessment and management of riparian habitats in the Central Valley (Table 9-5). One of these focal species is the bank swallow, which is discussed in detail in Chapter 7. The remaining species include various songbird species that historically occurred or are currently present in the Central Valley, plus one raptor, the Swainson's hawk which commonly nests in large riparian trees. Most of the RHJV focal species have experienced population declines from historical levels, and are currently considered to be special status species (see Table 9-5).

Research and monitoring studies of riparian songbirds in the Central Valley over the past decade have yielded a number of findings that help define the ecological needs of these species (Gardali et al. 2005, 2006 and Nur et al. 2005):

- the abundance of riparian-associated songbirds is associated with local-scale habitat and vegetation features
- species respond to general habitat characteristics (e.g., height of trees, shrub cover), but also species-specific vegetation composition
- tree height index and tree dbh index had a positive effect on species-specific abundance more often than other variables (i.e., size of tree is more predictive than tree cover or tree
- shrubs, particularly understory vegetation, are predictive of abundance (e.g., blackberry species richness) cover and mugwort cover)

- presence of water was found to have a negative effect as often as a positive effect; thus effect is indirect (i.e., mediated by vegetation); in some cases, availability of water can have a causal role, but it is generally not what birds are responding to most directly.

Table 9-6 summarizes the findings of Nur and colleagues (2005) regarding specific habitat associations of songbirds with various habitat features along the Sacramento River. One of the key findings of these recent studies (Gardali et al. 2005, 2006 and Nur et al. 2005) is that some species-specific responses to vegetation differ on a regional basis, emphasizing the need to base restoration plans on local data whenever possible. Another important finding is that some species will respond to a given habitat feature in a positive manner, while others are negatively associated with that same feature, and still others may show no significant association. The high coefficients of determination (R^2) in their statistical models indicate that birds respond to variation at home range or territory scale (i.e., within a few hundred meters). The study (Nur et al. 2005) was able to identify patterns in variation in abundance of 21 species, but could not identify ecological processes that created and maintained these patterns. Further ecological studies (for example, of predators and prey) and demographic studies are needed to understand the processes that drive the observed patterns.

Table 9-5. Status and habitat requirements of riparian bird species historically and currently breeding on the Sacramento (SACO) and San Joaquin Rivers (SAJO). Adapted from California Partners in Flight Riparian Bird Conservation Plan (CPIF and RHJV 1998) and Small et al. (1999).

Species	Statewide status	Historic Central Valley range	Current Central Valley status	Nest site	Habitat considerations
Bank Swallow *Riparia riparia*	*-CA Threatened* -Nesting populations appear to be declining	SACR	-50% of CA population breeds on Sac River	-Nests colonially -Digs burrows in friable soils vertical faces of bluffs or banks higher than 3.3 ft (1 m) tall -Dense nest colonies in eroded banks of Sac River	-Loss of nesting habitat from bank protection projects -Abrupt increases in water level during breeding season can cause nest failure. - River meander creates habitat
Black-headed Grosbeak *Pheuctecus melanocephalus*	*-No special status* -Population appears stable	SACR, SAJO	-Breeds on Sac and SJ Rivers -High winter survival but fairly poor nest success among Sac populations	-Highly variable. In riparian habitat, nests in willow, alder, box elder, and ash with high nest cover, 5–29.5 ft (1.5–9 m) from the ground	-Often nests in early successional habitat -Vulnerable to loss of riparian nesting habitat -Will nest on young riparian restoration sites on Sac River -Prefers semi-open canopy with moderate shrub cover and vertical stratification of vegetation layers
Blue Grosbeak *Guiraca caerulea*	*-CA Species of Special Concern* -Reduced in historic range	SACR, SAJO	-Breeds on Sac and SJ Rivers	-Nests in vertical forbs, young willows and cottonwoods, and herbaceous annuals, within 6.5 ft (2 m) of ground	-Prefers early successional habitat with annual forbs, young deciduous plants, and low canopy cover -Riparian edge species -Patch size and fragmentation seem unimportant

Species	Statewide status	Historic Central Valley range	Current Central Valley status	Nest site	Habitat considerations
Common Yellowthroat *Geothlypis trichas*	*-CA Species of Special Concern*	SACR, SAJO	-Breeds locally on Sac and SJ rivers, numbers probably greatly reduced due to loss of emergent wetlands	-Nests primarily in tall emergent wetland vegetation, nest on or near ground -Will nest secondarily in tall emergent upland vegetation	-Breeds in wetlands and open, early successional riparian areas -Spring/summer grazing reduces quality of nesting habitat
Least Bell's Vireo *Vireo bellii pusillus*	*-Federally Endangered* -Extirpated or reduced in much of historic range	SACR, SAJO	-Extirpated from Sac and SJ rivers -Potential for recolonization as range spreads northward, given appropriate habitat conditions	-Nests in dense shrubs, 0–6.5 ft (0–2 m) from the ground	-Prefers early successional riparian with dense shrub layer at nest height and a dense, layered canopy for foraging -Spring/summer grazing reduces quality of nesting habitat
Song Sparrow *Melospiza melodia*	*-No special status* -Breeding population greatly reduced in Sac River watershed	SACR, SAJO	-Breeding population extirpated from mainstem of Sac, occurs in nearby wetland management areas and on foothill tributaries -Nests on SJ River, very poor nest success	-Nests in shrubs and herbaceous vegetation, 0–6.5 ft (0–2 m) from the ground	-Breeds in early successional riparian, wetlands, and marshes -Responds quickly to restoration efforts, where adjacent source populations exist -Spring/summer grazing reduces quality of nesting habitat
Swainson's Hawk *Buteo swainsoni*	*-CA Threatened species* -In CA, may have declined up to 90%	SACR, SAJO	-Largest CA populations are between Sacramento and Modesto and in the northern San Joaquin Valley	-Nests in wide variety of tall trees	-Uses wide variety of open habitats with large nest trees, typically riparian forest or remnants -Disturbance can lead to nest abandonment -Poisoned by pesticides during migration and over winter

Species	**Statewide status**	**Historic Central Valley range**	**Current Central Valley status**	**Nest site**	**Habitat considerations**
Warbling Vireo *Vireo gilvus*	-*CA Species of Special Concern* -Declining in CA since 1979	SACR, SAJO	-Breeding populations extirpated on Sac and SJ rivers, some breeders on Sierra tribs	-Nests high in deciduous trees	-Prefers large deciduous trees associated with streams, semi-open canopy.
Willow Flycatcher *Empidonax traillii*	-Subspecies E.t. extimus *Federally Endangered* - All three CA subspecies are *CA Species of Special Concern*	SACR, SAJO -Central Valley subspecies was probably E.t. extimus	-No longer breeds on SJ or Sac rivers (Historic breeding status is unclear)	-Nests in willows, alders, and cottonwoods or other deciduous vegetation, 3–10 ft (1–3 m) from the ground	-Typically prefers dense patches and early successional riparian areas. -Spring/summer grazing reduces quality of nesting habitat
Yellow-breasted Chat *Icteria virens*	-*CA Species of Special Concern* -Reduced in much of historic range	SACR, SAJO	-Extirpated from the SJ -Breeds in low densities on Sac, somewhat higher densities on foothill tribs	-Nests in dense riparian shrubs and tangles of annual plants, within 10 ft (3 m) of ground	-Prefers dense, early successional vegetation, often among willow thickets -Spring/summer grazing reduces quality of nesting habitat
Yellow-billed Cuckoo *Coccycus americanus*	-*CA Endangered Species* -Much reduced in historic range, where it has not been extirpated	SACR, SAJO	-Extirpated from the SJ -Sac River hosts one of the major CA breeding populations	-Nests in willows, cottonwoods, box elders and occasionally orchards on the Sac River -Typically have high nest cover	-Large patch size is important -Benefits from healthy, meandering river system with intact hydraulics
Yellow Warbler *Dendroica petechia*	-*CA Species of Special Concern* -Extirpated or declining in much of breeding range	SACR, SAJO	-Extirpated from the SJ -Extirpated from mainstem of Sac River, with the exception of a few isolated pairs with poor nest success in 1999 -Breeds on Clear and Upper Mill creeks	-Nests in deciduous riparian plant species such as willows and cottonwoods	-Responds quickly to restoration efforts -Breeds in wet areas with early successional riparian communities, vertical stratification of vegetation, and a semi-open canopy -Spring/summer grazing reduces quality of nesting habitat

Table 9-6. Documented associations of Sacramento River songbirds and habitats. Source: Nur et al. 2005.

Species	Water present	Tree cover	Shrub cover	Herb cover	Tree sps richness	Shrub sps richness	Tree dbh index	Tree height index	Willow shrub cover	Willow tree cover	Cottonwood tree cover	Valley oak shrub cover	Valley oak tree cover	Blackberry cover	Mugwort cover	In model	R^2 (%)	# variables in model
Black-headed Grosbeak				-					+		+					yes	57	3
Blue Grosbeak							-					-	-			yes	26	3
Spotted Towhee	-	+	+		-		+									yes	45	5
Ash-throated Flycatcher						+		+								yes	30	2
Brown-headed Cowbird				+		-	+						-	+	+	yes	26	6
Red-winged Blackbird	-			+			-					+	-	+		yes	58	6
House Finch		-						+	-							no	18	3
House Wren		-	-	-				+	-	+	+		+			yes	49	8
Tree Swallow	+							+			+		-			no	29	4
American Goldfinch	-		-					+			-		-			yes	21	5
Song Sparrow			+					-				-				yes	71	3
Bushtit					+								+			yes	38	2
Western Kingbird	+	-	-								+			+		yes	46	5
Bewick's Wren				-			+							+		yes	44	3
American Robin	+		-	-			+					+		+		yes	29	6
Nuttall's Woodpecker				-				+								no	43	2
Bullock's oriole											+					no	2	1
Western Scrub-jay		+		+		+			+							yes	25	4
Lazuli Bunting	-			+					+						+	yes	55	4
Common Yellowthroat	-	+		+			+		+		-				+	yes	33	7
Western Wood-pewee	+		-	-		-	+								+	yes	77	6
Total +	4	3	2	5	1	2	6	6	4	1	5	2	2	5	4			
Total -	5	3	5	6	1	2	2	1	2	0	2	2	5	0	0			
Total	9	6	7	11	2	4	8	7	6	1	7	4	7	5	4			

- indicates negative association (significance level: $P < 0.10$)
+ indicates positive association (significance level: $P < 0.10$)
Note: grassland cover was tested but showed neither positive nor negative association with any of the species considered here.

9.5 Conceptual Model of Historical Conditions

Historically, riparian vegetation in California's Central Valley was typical of Mediterranean alluvial river systems, exhibiting non-equilibrium dynamics at the patch scale. The dynamic nature of Central Valley riparian zones was largely driven by annual flooding and a long summer drought (Thompson 1961). Annual flooding conferred a frequent disturbance regime via floodplain inundation, scour, and sediment deposition that maintained vegetation recruitment and survival. Water availability during summer drought was the primary gradient structuring vegetation establishment and distribution. This cycle of flooding and drought was, and is, especially significant to pioneer woody plant species, primarily Fremont cottonwood (*Populus fremontii*), and willows (*Salix* spp.), which rely on floods for bare seed beds, water, and nutrients, and which grow roots quickly to reach permanent water tables and a secure bank footing to resist subsequent floods (Braatne et al. 1996).

The dynamic ecological processes operating in the river-floodplain system were strongly affected by physical disturbance associated with annual flood-pulse cycles and interannual variability in climatic conditions that often created multiyear cycles of drought or floods. Under historical conditions, riparian vegetation in the Middle Sacramento River was extensive (see Figure 8-1). Topographic and landscape position affected the frequency of disturbance from floods (Figure 9-11), with sites at lower relative elevations and closer proximity to the main channel generally exposed to more frequent flooding. The more frequently disturbed sites on point bars in the main channel served as the primary recruitment zones for willows and cottonwoods, leading to development of early successional vegetation types such as willow/riparian scrub and cottonwood forest (Figures 9-10 through 9-14). Lower elevation sites along abandoned channels or oxbows were also more frequently disturbed by flooding and tended to serve as recruitment sites for cottonwoods and a different mix of willows, with Gooddings willow commonly occurring, creating young stands of cottonwood and willow and various types of mixed riparian forest. As scrub and forest stands on point bars and oxbow lakes and sloughs aged, they developed increased structure which interacted with subsequent flooding to increase hydraulic roughness and promote deposition of finer sediments. Over time, this interactive process of flooding and deposition increased the relative elevation of sites above the groundwater table and the summer baseflow level. This changed the physical environment, which combined with increased competition for light (and presumably nutrients and water) as the established trees and shrubs grew, created conditions that favored later successional species and vegetation types such as valley oak forest.

9.6 Effects of Changes on Cottonwood and Riparian Habitats

A variety of human land and water management actions have altered the extent and condition of riparian forests along the Sacramento River. Floodplain development for agriculture, habitat fragmentation, and river channelization and bank armoring (Figure 9-15) are examples of human alterations that have had large impacts on the riparian-floodplain ecosystem. The effects of these changes are discussed in detail in this section.

9.6.1 Harvesting/clearing of native riparian vegetation from floodplains

Forest clearing began with harvesting of trees along the river banks, primarily for fuel (Vaghti 2003, Vaghti and Greco in press, The Bay Institute 1998, Thompson 1961). In the mid- to late 1800s clearing of native riparian vegetation expanded to higher, less floodprone surfaces as farmers increased their use of the rich, alluvial soils to grow crops. These higher surfaces would have naturally supported valley oak forests and savanna, and native grasslands. Development of reclamation and flood control projects in the late 1800s through the mid-1900s (see Figure 2-1) allowed expansion of agriculture (row crops and to some degree orchards) into more intermediate elevation surfaces that formerly had been too floodprone for successful farming in many years. The net effect of these changes has been a vast reduction in the extent of riparian vegetation (Figure 9-16), to about 10% of historical levels from Red Bluff to Colusa and to less than 5% in most other parts of the Central Valley (compare Figure 8-1 with 8-4). Examination of the current patterns of distribution of these valley oak forests and grasslands show that they typically occupy higher relative elevation zones that overlap with the peak zones of development of orchards, and have greatly decreased from historical levels of abundance (see Figures 9-17 through 9-19).

Under current conditions, older floodplain surfaces (i.e., sites > 50 years old) that have not been converted to agriculture, still tend to support mixed riparian forest, while cottonwood forest occurs mainly on sites that are 20 to 50 years old, and riparian scrub and bare gravel and sand bars occupy sites that are < 20 year old (Figure 9-20, based the work of Greco et al, in review). There is very little valley oak forest/savanna or grassland left (Figure 9-17 and 9-18), presumably because they historically occurred on the higher relative elevation surfaces that have been converted to orchards and cropland (Figure 9-18).

9.6.2 Effects of flow regulation

As noted in Section 9.2, the Fremont cottonwood and other riparian vegetation species have evolved life history strategies that are closely tied to hydrologic parameters, including the magnitude, timing, and recessional characteristics of winter floods and spring snow-melt (Figure 9-1). Flow regulation on the Sacramento River has therefore had a number of potentially important effects on successional patterns of riparian vegetation. Below we discuss how changes in the frequency and magnitude of over bank flows have affected cottonwoods at high floodplain sites, and then consider how changes in the frequency, magnitude, and recession rates of seasonal floods have affected recruitment and establishment of cottonwoods along the mainstem.

9.6.2.1 Reduction in magnitude and frequency of overbank flows

Reductions in the magnitude and frequency of winter overbank flows in the post-dam era (Figure 8-3) have presumably led to an overall decrease in soil moisture during the growing season for cottonwoods and other riparian plants. We hypothesize that this has contributed to reduced growth rates and altered competitive interactions such that species more tolerant of somewhat drier conditions may become more dominant. This may have led to an increase in the abundance of box elder and walnut (see Wood 2003b, Vaghti 2003, Fremier 2003). This hypothesis is difficult to corroborate due to the absence of quantitative data on soil moisture and seedling survival in the pre- and post-dam eras. Nevertheless, it seems inevitable that soil moisture on the floodplain would be reduced due to a decrease in the frequency and magnitude of overbank flows, which we know has occurred as a result of flow regulation (Section 2.2.3). Further reductions in the frequency and duration of overbank flows could lead to a further shift towards species tolerant of even drier conditions, as has been documented in several rivers in the southwestern United States where the non-native, invasive saltcedar (*Tamarisk* spp.) has greatly expanded in abundance while native cottonwoods and willows have decreased (Shafroth et al. 2002).

Also difficult to quantitatively corroborate is the hypothesis that reduced frequency and magnitude of overbank flows has reduced floodplain sedimentation rates, such that the dynamics of riparian habitat succession have been altered. In general, a reduced rate of floodplain sedimentation should contribute to slower evolution of floodplain surfaces, which, for the most part, tend to increase in elevation over time. A slower rate of increase in floodplain elevations can be expected to contribute to slower succession towards Valley Oak, and other species that thrive in high, dry soils.

There is some indication that overbank flooding can be inhibited locally by meander bend cutoff, due to reductions in backwater effects associated with changes in planform curvature of the bend. In general, the more sinuous the bend, the greater the backwater effects are likely to be, and because cutoff processes generally reduce the sinuosity of bends, they generally promote reduced backwater effects (Harvey 1989).

Reduced high winter flow magnitudes may have also worked to locally reduce rates of progressive meander migration, which is generally accompanied by deposition of point bars, such that channel width remains roughly constant as the river migrates laterally across the floodplain. A reduced rate of point bar formation would tend over time to reduce the availability of fresh surfaces for establishment of seedbeds. However, as discussed in Chapter 3, the net change in progressive migration rates since the dams were constructed is not statistically significant. This appears to be due at least in part to the fact that there are many factors regulating progressive migration rates besides the magnitude and frequency of high winter flows.

The fact that progressive migration rates have remained stable despite the effects of flow regulation (Micheli and Larsen in preparation; Table 3-4) stands in contrast to the observation that the rate of production of new land has decreased by a factor of two (Greco et al. in press), from 398 ac yr^{-1} (161 ha yr^{-1}) before the dams were built to 200 ac yr^{-1} (81 ha yr^{-1}) in the post-dam era. The discrepancy between progressive migration rates and rates of production of new land may be due to one or more of the following factors:

1. Migration rates are inferred from changes in channel centerlines from one series of aerial photos to the next, whereas rates of production of new land were inferred from changes in the position of channel boundaries under "low-water" conditions. To the extent that channel boundaries depend sensitively on flow, the analysis of rates of production of new land will be subject to uncertainties introduced by differences in flow from one photo series to the next, whereas the analysis of channel center lines would be relatively robust against such variability.
2. The decrease in rates of production of new land may reflect a system-wide change in channel geometry. For example, if point bars have become systematically steeper in the post-dam era, then the rate of production of new land could decrease even as progressive migration rates have remained stable; for a given increment of bank erosion, the area of point bar required to maintain a stable channel width when point bars are steep can be smaller than it needs to be for more gently sloping point bars.
3. The decrease in rates of production of new land could also reflect the increase in the frequency of partial cutoffs relative to complete cutoffs in the post-dam era, with partial cutoffs contributing less new land per event, because they are shorter.

The discrepancy between observed migration rates and observed rates of production of new land along the Sacramento River is important to understand and should be the subject of further work and a review of the modeling assumptions that create these results.

The increased incidence of meander bend cutoffs in the post-dam era (as identified in Chapter 3) suggests a trend toward straighter reaches, which would tend to have lower progressive migration rates. If this is the case, then the channel may be evolving toward a state in which point bar formation (and the formation of suitable seedbeds) is reduced relative to pre-dam conditions. However the effects of reduced point bar formation on cottonwood recruitment and establishment would be offset to some extent by increased recruitment associated with the increase in cutoff, which has been shown to provide an important alternate pathway for cottonwood establishment (Greco et al. in press). The increase in cutoff frequency is presumably related to an increase in floodplain erodibility, as discussed in Chapter 3 and again below in Section 9.6.3.

9.6.2.2 Changes in timing and magnitude of flows important for recruitment and establishment

By altering conditions for seed recruitment and establishment, changes in the flow régime have affected Fremont cottonwoods near the river as well as those on higher floodplain surfaces. For example, as discussed in Chapter 2 the magnitude of peak winter flows and spring flows has been reduced in the post-dam era. This would tend on average to reduce the amount of area scoured bare for viable seedbeds. However, it appears that appropriate seedbed is still fairly common along the middle Sacramento River based on the observations reported by Roberts et al. (2002), TNC (2003), Wood (2003a), Morgan (2005), and Morgan and Henderson (2005ab).

The reduced magnitude, and possibly altered timing, of spring flows may have also affected cottonwoods by encouraging recruitment on low depositional surfaces that become inundated by subsequent winter floods or by elevated summer baseflows (Morgan 2005, Morgan and Henderson 2005ab). Despite the high tolerance of Fremont cottonwood seedlings to extended periods of inundation, inundation and mortality of seedlings by summer base flows has been observed on the Sacramento River (Morgan 2005, Morgan and Henderson 2005ab). By keeping water tables high during summer, elevated summer base flows may also contribute to reduced elongation of roots and thus increased susceptibility to scour in winter floods. On the other hand, alterations to the spring snowmelt hydrograph (CDWR 2001) may have contributed to a reduced frequency of successful recruitment events (Roberts et al. 2002, TNC 2003a), with flow recession rates in the post-dam era being so rapid that decreases in soil moisture in seedbeds often outpace the rate of root growth, and thus lead to high mortality during seedling initiation. Taken together, these factors could over time limit the success of initiation and establishment of cottonwoods.

Figure 9-21 summarizes the results of flow recurrence interval (RI) analysis conducted by Lowney and Greco (2003) which indicate that flow of about 160,000 cfs historically occurred at a 5-year RI during 1879–1943 prior to construction of Shasta Dam. Under current flow management conditions (post-Shasta Dam, 1944–2000) this type of flow event occurs much less frequently, with a recurrence interval of approximately 20 years. It is likely that less frequent recruitment of cottonwoods and riparian forests may result from such hydrologic alterations (Roberts et al. 2002).

Morgan (2005) concluded that there were three primary attributes of the current altered hydrograph that limit cottonwood seedling survival at the middle Sacramento River study sites: 1) the reversal of summer flows such that there is now an increasing trend in summer flow levels during cottonwood seed release and seed germination periods, 2) rapid stage declines during the spring pulse flow such that root growth in seedlings established during the typical recruitment period cannot keep up with declining water levels, and 3) the immediate drop in stage late in growing season when reservoir releases for summer irrigation cease.

Reduced abundance and growth rates of seedlings and unsuitable conditions for seedling establishment have been reported on many creeks and rivers in the western US affected by flow regulation (e.g., Rood and Mahoney 1990 and 2000, Braatne et al. 1996, Friedman et al. 1998, Shafroth et al. 2002, Rood et al 2003 and 2005, McBain and Trush 2002, Stella 2005, Stillwater Sciences 2006). Examples include the Rio Grande in New Mexico, the Salt and other rivers of Arizona, and Bishop Creek and the Sacramento, Owens, Tuolumne, San Joaquin, and Colorado rivers in California.

9.6.3 Effects of changes in erodibility of channel banks and the surrounding floodplain

In addition to being affected by changes in flow régime, the habitats of Fremont cottonwood and other riparian vegetation species along the Sacramento River have also been affected by changes in the erodibility of banks and floodplains. The relative importance of changes in flow and erodibility at any given point along the river depend on local conditions. We present several generalizations about potential effects of changes in erodibility below.

9.6.3.1 Bank armoring

As discussed in Chapter 3, bank armoring or revetment activities along the Sacramento River have resulted in substantial local reductions in rates of channel migration. By stopping channel migration, bank armoring also locally halts production of new surfaces for seedling recruitment. The progressive increase in bank protection over the last century has presumably contributed to the observed decrease in the overall rate of production of new land in the riparian corridor of the Sacramento River (Greco et al. in review). Bank armoring activities also result in direct loss of vegetation immediately adjacent to the river and thus affect the overall abundance of Fremont cottonwood in riparian corridor. By armoring banks against erosion, installations of riprap and concrete have also presumably reduced the propensity of channels to cutoff, and thus reduced cottonwood recruitment via that successional pathway.

Bank armoring also appears to affect adjacent point bar topography, with a general steepening of the point bar cross-sectional slope near revetted banks (Buer 1994; Buer et al. 1989). Wood (2003a) provides comparisons of cross-sections for point bars with and without bank armoring immediately upstream that demonstrate this topographic effect, which results in decreased potential seedbed area for cottonwood recruitment and likely reduced the frequency of successful seedling establishment because the steeper bar slope is associated with more rapid stage declines per unit flow decrease (see Section 9.6.4).

9.6.3.2 Levee constraint of the channel meander zone

The installation of levees along the Sacramento River has also presumably affected the extent and health of riparian forests (mainly middle and late successional forests) by causing direct losses in habitat area both at the levee site, and also behind the levees where land is usually converted for agricultural and other uses. Levee construction reduces the area subject to migration, cutoff, and flood pulses, and thus affects the dynamism of the floodplain forming processes which create new surfaces for cottonwood recruitment and the flood-pulse cycle of inundation and fine sediment deposition that drives riparian successional processes.

9.6.3.3 Clearing of riparian forests

As noted in Chapter 3, the clearing of riparian forests from the floodplain (discussed in section 9.6.1 as a direct effect on cottonwood populations) has apparently worked to increase rates of progressive migration and meander bend cutoff over the last 100 years (Micheli and Larsen in prep). This would tend to increase production of surfaces for recruitment of riparian vegetation, however this potential benefit is countered by the hydrologic alterations that greatly reduce the frequency of successful cottonwood recruitment. However, as discussed in greater length in Chapter 3, the effects of reduced floodplain erodibility may have been offset to some extent by the effects of reduced frequency and magnitude of high winter flows. Even so, cutoff frequency in the post-dam era is higher than it was in the pre-dam interval. This implies that recruitment via the cutoff processes pathway may have accelerated somewhat in the post-dam era. This would tend to offset, to some extent, the vegetation losses and reductions in recruitment associated with bank protection and the levee construction.

9.6.4 Changes in cross-sectional geometry

By affecting the delivery of flow and sediment and by affecting the way flow interacts with channel banks and floodplain area, management of the Sacramento River over the last 160 years may have led to substantial changes in cross-sectional geometry. The extent to which this is the case has probably varied significantly as a function of local conditions. For example, it has been shown that banks that have been armored with concrete and riprap are associated with relatively narrow, deep channels, compared to banks that are free to migrate (Buer 1994; Buer et al. 1989). This suggests that revetment may have lead to localized narrowing and deepening of channels, presumably due to constriction of the meander zone and associated increases in shear stress on the bed (see Chapter 3). Meander zone constriction would also probably result from construction of levees in the immediate vicinity of the channel. However, changes in cross-sectional geometry associated with levee construction have not been quantitatively documented, and probably vary on a case-by-case basis depending on local conditions (i.e., levee setback distance and planform geometry of the river).

Management-related changes in cross-sectional geometry are important to understand, because they are likely to have important implications for cottonwood and other riparian vegetation. For example, a narrowing and deepening of channels (which apparently can result from bank protection installations) would be accompanied by a steepening of the point bar, and thus a decrease in the area of the potential recruitment zone (Wood 2003a). The steeper point bar would also support more efficient draining of soil moisture during recessional flows. Narrowing and steepening could also change the local stage-discharge relationship, such that a given increase in discharge produces a larger increase in stage, leading to a greater relative elevation of recruitment relative to natural conditions. A steeper stage-discharge relationship could also contribute to faster decreases in stage during the receding limb of the hydrograph. Deepening and narrowing of cross sections probably also leads to a decrease in the average water-surface elevation of the river. We expect this would be accompanied by a local decrease in the elevation of the groundwater table, and reduced success of seedling initiation and establishment in the immediate vicinity of the river. More distal sites would also presumably be affected by decreased water-surface elevations on the river, due to a decoupling of the mainstem from the surrounding floodplain, and the resulting increased incidence of desiccation-related mortality of newly established recruits on high surfaces.

Overall, we expect that newly initiated seedbeds in deeper, narrower reaches are smaller in area and are subject to faster desiccation (i.e., with lower frequencies of successful recruitment) compared to broader shallower reaches. Hence, if management activities have produced an increased abundance of deep, narrow reaches, they may have contributed to an overall decline in cottonwood recruitment.

9.6.5 Introduction of non-native plant species

Exotic plant species can alter the structure and dynamics of natural ecosystems. Non-native plant species can impact native wildlife by displacing native vegetation that is used for nesting or as a food source. Once established, non-native plant species can alter nutrient cycling, energy fixing, food web interactions, and fire and other disturbance regimes to such an extent that the native landscape is changed. Habitat fragmentation contributes to the spread of non-native species by increasing edge habitat, which provides greater opportunities for invasion by exotic species (Cox 1999). Ecosystem alterations resulting from non-native plant species invasions can be exacerbated by activities such as grazing and vegetation clearing that create favorable conditions for further non-native plant establishment (Cox 1999, Randall and Hoshovsky 2000). Alteration of historical flooding regimes by flow regulation further promotes invasions by non-native species by eliminating processes necessary for recruiting and maintaining native plant species (Cox 1999).

A number of non-native invasive species occur along the Sacramento River. The most common species are giant cane (*Arundo donax*), edible fig (*Ficus carica*), Himalaya berry (*Rubus discolor*), and tree of heaven (*Ailanthus altissima*) (Wood 2003b, Vaghti 2003, Vaghti and Greco in press). Saltcedar (*Tamarix* spp.), walnut (*Juglans*) hybrids, and *Paulownia tomentosa*, are additional non-native species of potential concern along the middle Sacramento River. A large number of non-native forbs and grasses also occur throughout the riparian corridor on point bars and floodplain surfaces (Vaghti and Greco in press).

9.7 Conceptual Model of Current Conditions

Flow regulation, land clearance, bank armouring, and levee construction have had direct and cumulative adverse impacts on native riparian plant communities along the middle Sacramento River (see Section 9.7.1 and Table 9-7). Alterations in fluvial geomorphic processes, the primary drivers affecting the composition and structure of riparian vegetation, have affected both the initiation and establishment of pioneer species such as Fremont cottonwood and various willow species and the subsequent trajectories of plant community succession. Many existing stands of cottonwood forest will transition to box elder and walnut dominated stands over the next several decades instead of the historical mid-seral mixed riparian forest stands that were co-dominated by Oregon ash, sycamore, cottonwood, and valley oak (Fremier 2003, Vaghti 2003, Wood 2003b) (Figure 9-20). These new stands of box elder and walnut mid-seral forests are unlikely to transition to valley oak riparian forest, which was the dominant later seral type historically, given the current lack of valley oak recruitment throughout the riparian corridor (Vaghti 2003, Wood 2003b).

Valley oak and sycamore dominated stands are greatly reduced from historical levels, due largely to conversion of mid to high level floodplain surfaces to agriculture, particularly orchards and other disturbed cover types (Figures 9-18 and 9-19), coupled with alterations in the natural flood-pulse regime that appear to favor box elder and walnut over valley oak, sycamore, and Oregon ash as the dominants in mid-seral forests (Fremier 2003, Vaghti 2003, Wood 2003b). Recruitment of valley oak and sycamore, in particular, appears to be nearly nonexistent under current conditions. Recruitment of box elder is common on younger to medium-aged floodplain surfaces, while recruitment of walnut is common on older floodplain surfaces (Fremier 2003, Vaghti 2003, Wood 2003b).

9.7.1 Potential limiting factors

Existing information in the literature on riparian vegetation dynamics and cottonwood life history requirements identifies a wide range of factors, both natural and anthropogenic, that have the potential to limit cottonwood populations. The list of potential limiting factors can be used as an initial list of hypotheses about factors that may be impairing cottonwood populations in a given area. The list of hypotheses can then be refined using existing information on historical and current site conditions in a particular study area, such as the Sacramento River corridor between Colusa and Red Bluff, to identify the factor or factors most likely to be limiting cottonwoods under current conditions and point out where further study might be required to reduce key uncertainties related to potential management actions (see Section 9.8). Table 9-7 summarizes our current understanding of the relative importance of various factors that have the potential to limit recruitment and survival of Fremont cottonwood along the middle Sacramento River.

Historically, impacts caused by widespread land clearance and tree harvesting (for fuel for early steamboats) were probably the most important limiting factor in the pre-Shasta Dam period from the mid-1800s through the mid-1900s. Subsequent to the completion of Shasta Dam, and continuing into the present day, the most important factors limiting cottonwood recruitment are hydrologic and geomorphic alterations in general, and specifically alteration in spring snowmelt peak flow/flood pulse timing and summer baseflow regime coupled with restriction of channel migration by bank revetment and levees (Table 9-7).

9.7.2 Restoration implications

A variety of conceptual models may be required to adequately address riparian restoration needs under current conditions. For example, contemporary flow regimes on managed rivers, combined with multiple land and water uses, will likely constrain the amount of channel migration or avulsion that might be expected under restoration plan implementation. Under such conditions, creation of new point bar surfaces or abandoned channel sites suitable for natural colonization by cottonwoods and willows may be limited or under-utilized where it does occur. Periodic resetting of successional processes to maintain a diverse mosaic of vegetation and habitat types may thus require alternative types of natural disturbances (e.g., fire, disease, windthrow, or extreme floods) or human intervention (e.g., mechanical removal of vegetation and ground disturbance to create bare substrates suitable for recruitment, or revegetation using horticultural techniques, prescribed fire, control of invasive plant species).

Table 9-7. Summary of current understanding of factors potentially limiting Fremont cottonwood recruitment and survival along the Sacramento River. For each potential limiting factor, a brief description is provided, along with expected results (indicators) that would be observed if that factor is indeed limiting, and a brief summary of our current understanding regarding the importance of each factor along the middle Sacramento River. Management priority ratings are provided to indicate suggested prioritization for targeted management actions.

Potential limiting factors	Expected result if factor is limiting	Current understanding	Management priority
1. Hydrologic alteration			
Seedbed limitation. Decreased magnitude of peak flows results in poor seedling recruitment because of decreased overbank flooding, scour, and sediment delivery to the floodplain.	*Lower winter peak flows *Heavily vegetated floodplains and lack of recent sedimentation	Winter and spring peaks are reduced in magnitude and frequency, reducing level of scour and deposition that creates seedbeds; has potential to be a limiting factor at some sites, and could become more important if further reductions in peak flows occurred.	Medium - Low
Out-of-phase timing. Changes to the timing of peak and sustained high flows prohibit seedlings from establishing on floodplain surfaces.	*Lower flow peaks and duration during the spring seed release period *Recession rates too rapid for seedling initiation and establishment in suitable sites *Seedlings initiate on low relative elevation surfaces but do not survive because such surfaces are prone to scour during winter peak flows	Spring snowmelt pulse magnitude, timing/duration, and recession rate affected by regulated conditions; likely to be a primary limiting factor.	High
Intra-annual survival. Increased summer base flow and sharp drop at end of summer desiccates seedling roots.	*Evidence of high seedling mortality *Shallow rooting depth on existing seedlings *Altered summer flow patterns resulting in rapid stage declines	Increased summer baseflows followed by a sharp flow drop after the main irrigation season may desiccate seedling roots; high probability under post-dam flow regime, but could be addressed by implementing a more gradual ramp down at end of irrigation season.	High
2. Geomorphic alteration			
Channel morphology. Channel incision and/or aggradation since flow regulation resulting in floodplain elevations inappropriate for cottonwood regeneration.	*Channel incision below root systems of existing trees *Extreme floodplain aggradation since establishment of existing trees	Not fully explored, but available evidence does not suggest this is a widespread problem.	Low
Sediment supply. Reduced sediment delivery to the floodplain prevents development of appropriate seedbeds and floodplain aggradation.	*Reduced rates of floodplain aggradation *Lack of fine sediment deposits on floodplain surfaces after flood events	Sediment from upper watershed is trapped by Shasta Dam, but tributary inputs and local erosion processes appear to provide adequate fine sediment in most reaches.	Low
Channel migration. Reduced channel migration prevents the creation of new seedbed sites.	*Reduced rates of channel migration *Lack of potential recruitment sites	Levees and bank revetment affect channel migration processes. Steepening of point bars adjacent to	High

Potential limiting factors	Expected result if factor is limiting	Current understanding	Management priority
		armored banks may reduce both areal extent and probability of successful recruitment.	
3. Land clearance			
Removal of native vegetation for agriculture, roads, levees, bank revetment, and other human development.	*Direct loss of native riparian vegetation and removal of potential seedling recruitment sites	Historically this has been a key limiting factor. Current management practices and regulations should help reduce the importance of this factor in the future.	Medium - Low
4. Herbivory effects			
Livestock. Grazing by cattle and other livestock results in high seedling mortality.	*Reduced seedling survival and end-of-summer residual dry matter in unfenced grazed areas relative to fenced plots and other reference sites	Does not appear to be a primary factor along the mainstem Sacramento River, but may be important locally.	Low
Native herbivores. Grazing by beaver, deer, or other native herbivores limit seedling survival.	*Grazing damage to seedlings, saplings, and adult trees at reference sites	Minimal concern over beaver damage of planted cottonwoods at restoration sites. Non-native grasses and weeds and reduced frequency of flooding may promote larger rodent populations, which can result in herbivory on cottonwood seedlings or saplings, particularly at higher elevation restoration sites. No evidence of major problem at potential point bar recruitment sites. Long-term control of native herbivores is not likely practical, but short-term control might be feasible to facilitate one or more cohorts of natural cottonwood recruitment.	Low
5. Effects of other factors			
Nutrient enrichment. Nutrient enrichment can increase competition from herbaceous vegetation, particularly many weed species, decreasing growth and survival of cottonwood seedlings.	*Elevated nitrogen levels in floodplain soils	Unknown, but nutrient enrichment may occur in some potential recruitment sites, particularly abandoned channels, sloughs and oxbow lakes adjacent to agricultural lands.	Medium - Low
Soil compaction. Soil compaction from earth moving has negative affects on seedling recruitment.	*Physical disturbance from earth moving and other activities decreases survival of cottonwood seedlings	Unknown, but current practices may not contribute to soil compaction except in very localized areas.	Low
Recreation. Recreational uses change the disturbance level, microclimate, competition, or other environmental factors that negatively affect cottonwood seedlings and saplings.	*Reduced seedling and sapling density and increased damage to existing trees in heavily-used recreational areas	Unknown. Appears unlikely to be a widespread problem but trampling may be a limiting factor along shorelines heavily used for fishing, swimming, and boating.	Low
Competition. Existing vegetation on potential seedbed areas excludes cottonwood establishment.	* Heavy early season vegetation on banks and sandbars *No bare substrates available during the cottonwood seed release period	Generally unknown, elevated summer baseflows potentially favor herbaceous species.	Medium

Potential limiting factors	Expected result if factor is limiting	Current understanding	Management priority
Seed availability. Seed sources are inadequate to sustain cottonwood stands.	*Few or no female trees present *Heavily skewed sex ratio favoring male trees	Unknown: no data on sex of trees in the area. Seems unlikely to be a widespread problem. Simple reconnaissance during expected seed release period could address the relative abundance of mature females producing seed. May be of more importance in subreaches that have been more heavily disturbed.	Medium
Disease. Cottonwood stands are limited by insect and/or microbial infestations.	*High numbers of dying young and adult trees *Evidence of pathogen damage to young and adult trees	Unknown, no disease or pests cited in background documents; considered a low probability limiting factor.	Low

9.8 Management Implications

9.8.1 Actions to consider

9.8.1.1 Manage recession limb of high flow events in wet water years to promote seedling establishment

Widespread flow regulation along rivers in the Sacramento Valley (and throughout the Central Valley) contributes to the decline of pioneer riparian vegetation populations, particularly Fremont cottonwood, but also provides a critical opportunity for their recovery. In light of our knowledge of the life history timing of sensitive species and the environmental conditions required during key life phases, flow releases may be designed efficiently and implemented strategically to enhance these species' habitat conditions and population trajectories. Managed flow releases have already been successfully applied for restoration of riparian trees elsewhere (e.g., Rood and Mahoney 2000, Rood et al. 2003, Rood et al. 2005).

In order to promote enhanced process-based restoration of woody riparian vegetation along the Sacramento River, managed releases (which we call here 'recruitment flows') will need to mimic natural hydrographs in several key ways. In particular, recruitment flows will need to reach a peak discharge during the peak seed release period to saturate seedbeds and induce seed germination at elevations suitable for long-term establishment. The necessary peak discharges are well within the active channel and require no overbank flooding. Following peak discharge, the flow recession rate will need to be sufficiently gradual to ensure that seedling roots maintain contact with the water table and capillary fringe as they grow. Because seed production is abundant every year (Stella 2005; Stillwater Sciences 2006: Stella et al., in press), restoration flows will likely be needed only in years with high total runoff volume in order to recruit younger cohorts into tree populations.

Various models that apply the recruitment box model to specific rivers have been developed to guide flow management actions designed to promote cottonwood recruitment. For example, Stillwater Sciences (2003) developed and applied such a model to predict maximum potential recruitment of Fremont cottonwood that might occur under different flow management scenarios for a 150-mi (241-km) reach of the San Joaquin River below Friant Dam. Another similar application, the TARGETS model, was developed for application to black cottonwood establishment on the Trinity River (Alexander 2004). An experimental recruitment flow designed with guidance from the TARGETS model is being implemented in Spring 2006 (J. Bair, pers. comm., 2006). Stella and colleagues (Stella 2005, Stillwater Sciences 2006) recently calibrated and tested a similar but different recruitment box model on the lower Tuolumne River.

Recruitment flow targets related to the timing and recession rates of spring pulse flows, as determined by the TNC and CDWR recruitment box calibration studies for cottonwood seedling initiation and establishment, are a component of the SacEFT being developed by ESSA in collaboration with TNC and Stillwater Sciences (ESSA 2005, 2006). The SacEFT is designed to be a decision analysis support tool that can help water and resource managers explore the ecological effects of different flow regimes and other management scenarios. The recent studies by TNC (2003) and DWR (Morgan and Henderson 2005b) of cottonwood recruitment processes along the middle Sacramento River have confirmed the validity of the recruitment box modeling approach and have calibrated various parameters for specific application to the Sacramento River.

These studies provide sufficient information to develop an experimental spring flow release to promote cottonwood recruitment in the middle river. The TNC (2003) study presents several recruitment flow scenarios that demonstrate that flows could be managed during wetter years to promote successful cottonwood recruitment without requiring any increase in the total volume of water released during May and June (Figures 9-22). Successful recruitment flows at the middle Sacramento River study sites tend to mimic the natural rates of recession of inflow to Shasta Reservoir during some wet years (Figure 9-23). These findings and those from other studies in western rivers (Rood et al. 2003 and 2005, Stella 2005, Stillwater Sciences 2006) suggest that managing flows to promote cottonwood recruitment during wet years is a viable restoration strategy, and one that has many ancillary benefits to other target species (see below).

Concurrent monitoring of seedling initiation and establishment could be used to track the success of such an experiment flow release, and fine-tune the details for subsequent application. A similar program is already underway on the Trinity River where high water conditions in Spring 2006 allowed operators to release an experimental recruitment flow targeted at promoting cottonwood seeding recruitment at a specific restoration site. Results of the monitoring currently underway should be highly informative for any similar efforts that might be conducted in the Sacramento River system.

Indexing Flow Planning to Water Year Type

The volume of water available for a recruitment flow (and therefore the range of potential magnitude, duration, and flow recession) will be largely determined by contemporary hydrologic conditions. Recognizing the stochastic nature of historical floods as well as the extremes of interannual water availability within California's climate, we need to take advantage of years when surface water is abundant to optimize recruitment.

Water year classification systems provide a means to assess the amount of water originating in a basin. Various classification schemes are currently used in water resource planning and river restoration plans to index water years based on precipitation volume and measured or predicted runoff (e.g., McBain and Trush 2000, Stillwater Sciences 2003, CDWR 2005). Currently, the California Department of Water Resources (CDWR) uses a water year classification developed by the State Water Resources Control Board for the Sacramento and San Joaquin rivers that calculates a weighed annual index based on three water projections: the current water year's unimpaired fall and winter runoff forecast (October–March), the current spring (April–July) unimpaired runoff forecast; and the previous water year's index (CDWR 2005). The indices for both rivers define one "wet" classification, two "normal" classifications (above and below normal), and two "dry" classifications (dry and critical), for a total of five water year types.

The ideal condition for promoting tree recruitment is to release a relatively large flow (but one that is still within the active or bankfull channel) in a wet water year when upstream reservoirs are fairly full (from previous wet or above-normal years). Under these conditions the flow pulse can be sustained to allow moist conditions to persist at fairly high relative elevations on floodplains until seedlings can grow extensive root systems and reach the perennial water table. Under less ideal conditions (e.g., above-normal years), lower magnitude flows can be used to encourage recruitment on lower floodplain and bank surfaces. Natural variability in the timing, magnitude, and duration of spring pulse flows helped to establish and maintain a diverse and dynamic mosaic of native riparian plant species and vegetation types. Managed flow regimes can be varied to some degree from year to year to help restore and maintain vegetation diversity (see "interannual flow planning" below for further discussion).

Recruitment flows would not normally be targeted for normal, dry, or critically dry water years, since reservoir volumes would not likely be sufficient to meet recruitment flow needs. In these years, the limited water supply should be conserved to meet human needs and to provide stable baseflows for recharging water tables in late summer, when existing trees would be most vulnerable to drought mortality. Because of these considerations, we recommend a dual approach to flow management for riparian vegetation issues (Table 9-8): (1) for wet and above-normal years, a focus on seedling recruitment; and (2) in all other years, a focus on survival of seedling and sapling cohorts recruited in previous years by sustaining groundwater levels in summer.

Table 9-8. Primary riparian flow management objectives, by water year type.

Water year type	Approx. percent of years	Management objectives	Desired frequency of recruitment flows
Wet and above-normal	40%	Spring *recruitment flows* to establish seedlings on appropriate seedbed surfaces, with summer flow conditions sufficient to maintain seedlings on desired surfaces.	Target is an average of 1 or 2 cottonwood cohorts successfully established each decade, which on average would require a recruitment flow in 25–50% of wet and above-normal water years.
Below-normal, dry, and critically dry	60%	No planned recruitment. Need to maintain summer water table for young cohorts to become established (*maintenance flows*).	NA

Though indexing restoration flows to annual flow volumes is an important management consideration, for this approach to be successful it must result in a sustainable population structure for pioneer riparian trees. Most willow and cottonwood trees have short life spans, typically less than 80 years in the case of the three most common pioneer species in Central Valley riparian zones: Fremont cottonwood, Goodding's black willow, and narrow-leaved willow (Stella 2005). For these species' populations to be sustainable, new cohorts of these trees need to be created at short enough intervals to replace adults killed by disturbance or senescence. Research on riparian forest stands in western North America indicates that successful recruitment events on meandering alluvial rivers typically occur after flows representing a 5- to 10-year recurrence interval (Bradley and Smith 1986, Cordes 1991, Reid 1991, Howe and Knopf 1991, Stromberg et al. 1991, Stromberg et al. 1993, Rood et al. 1997, Scott et al. 1997, Cordes et al. 1997, Rood et al. 1998), although some studies cite intervals as short as 3 years (Baker 1990, Howe and Knopf 1991) and others as long as 30 to 50 years along some non-meandering and confined rivers (Hughes 1994). When lateral migration is prevented by natural geologic constraints or by human channel stabilization efforts (e.g., through use of bank revetment and levees), flood deposition may be the only fluvial process available to create sites suitable for cottonwood establishment (Friedman et al. 1997). The result of such channel confinement is less frequent establishment of cottonwoods and a narrower riparian forest zone (Friedman et al. 1997). In addition, if water management reduces the variation in annual peak flows, cottonwood establishment and regeneration of riparian forests may decrease (Friedman et al. 1997).

Recruitment flows on the middle Sacramento River should be initially implemented on the lower end of that frequency range (i.e., 1 or 2 recruitment flows per decade) to ensure establishment of viable seedling cohorts in the short-term and to account for potential large-scale mortality events due to desiccation during drought years or high levels of scour and inundation during extremely wet years. However, as discussed above, the potential for a recruitment flow in any particular year will be constrained to a large degree by factors such as the magnitude and timing of winter rainfall, prior year hydrology, and reservoir operational issues.

Spring recruitment flows

For the purposes of this project, recruitment flows refer to controlled releases that occur during the spring seed release and dispersal period for pioneer riparian trees (particularly Fremont cottonwood, but managers may also want to promote recruitment of Gooding's black willow, the other dominant pioneer tree species along the middle Sacramento River corridor). These flows are designed to mimic the historical snowmelt pulse. In mid-spring, a managed release would occur, characterized by a sharp increase in flow to a maximum, then a more gradual recession rate throughout the late spring and a return to baseflow in early summer. Commonly, flow regimes are characterized as to their seasonal timing, magnitude, duration, and rate of change in discharge or river stage. The biological importance of each of these flow measures relative to pioneer riparian tree recruitment is summarized briefly below.

> *Timing*. For cottonwoods (and willows), whose seeds are viable only for several weeks, seed release must coincide with wet conditions and seedbed availability to produce a successful cohort. Appropriate flow timing is therefore the first condition necessary for a successful recruitment flow, and constraining flood timing will conceivably benefit some species over others. The annual chronological order of spring seed-releasing pioneer species along the middle Sacramento River and most other Central Valley rivers is: arroyo willow, Fremont cottonwood, Goodding's black willow, and narrow-leaved willow (Kondolf et al. 2000, Peterson 2002, McBain & Trush 2002, Stillwater Sciences 2006, Stella et al. in press). Recruitment flows should be targeted from late April to early June to improve cottonwood recruitment, and late May to early July to benefit Goodding's black willow. In regions where daily air temperature data are recorded, a degree-day model may be used to improve prediction of the seed release period and refine the timing of recruitment flows from year to year to promote maximum water-use efficiency (Stella 2005, Stillwater Sciences 2006, Stella et al. in press, Morgan and Henderson 2005b). Flows prior to mid-April will likely miss the seed release window for these species (but may benefit arroyo willow), and later flows will likely benefit Goodding's black willow and particularly narrow-leaved willow, which releases seeds throughout most of the summer. This latter species also likely benefits from elevated summer baseflows in the absence of spring peaks because of its vigorous sprouting ability.
>
> *Magnitude*. The magnitude of a spring flow pulse determines how high on the banks and floodplain the river stage reaches, and therefore, how high and extensive the areas of potential recruitment are. Willow and cottonwood seedbeds need to have bare, moist, fine-grained mineral substrates for germination to be successful. Besides topography, the annual extent of recruitable area is influenced by the magnitude of scouring winter flows (discussed below) and the sediment supply and dynamics. At present, target elevations based on the site-specific results from the three point bars studies by TNC and CDWR (TNC 2003a, Morgan and Henderson 2005b) can be used to determine the magnitude of recruitment flows needed to establish cottonwood seedlings on the target surfaces. Calibration of the recruitment box model to additional sites along the Sacramento River would facilitate a corridor-scale planning process, in which discharge targets for optimizing seedbed inundation during recruitment flows could be identified using GIS-based hydraulic and digital terrain models (Stillwater Sciences 2003, 2006). These discharge targets would determine the

desired peak discharge reached at the beginning of the recruitment flow, subject to flood control and dam operational constraints, to optimize successful seedling initiation and establishment within the desired relative elevation target zone. Currently, study sites on the middle Sacramento River are located about 10 mi (16 km) apart (at RM 192, 183, and 172) and likely provide an accurate characterization of conditions within that reach or sub-reach. Until further studies are conducted, extrapolation of the TNC and CDWR findings to other locations along the river can be accomplished using the relative elevation model developed by Greco and colleagues (in press) as a surrogate for more detailed, site-specific hydraulic modeling and development of empirical stage-discharge relationships required for recruitment box model application.

Duration. Recruitment flow peaks should be of sufficient duration to fully saturate the seedbed substrate (down to the perennial water table) and allow for floating seeds to raft up onto floodplain surfaces. Most willow and cottonwood seeds germinate within 24–48 hours after wetting (Pelzman 1973; Guilloy-Froget et al. 2002; Stella et al. in press) and have an initial period slower root growth that may last a week or more (Stella 2005, Stillwater Sciences 2006), so flows likely need to be maintained at fairly constant levels several days to a week (maximum) at peak levels to induce germination on the desired geomorphic surfaces. Since brief flow peaks will limit the quantity of seeds rafted onto floodplains from upstream areas, recruitment flows should occur during peak seed release, when waterborne seed density is highest, to most efficiently collect rafted seeds. The restoration hydrograph should be designed to maintain peak flow for several days to a week followed by a gradual initial ramp down in order to concentrate and deposit these seeds at appropriate higher elevation surfaces within the target zone (creating a “shoulder” or “bench” on the recession limb of the recruitment flow hydrograph. With shorter peak flow duration and more rapid ramp down rates, seed deposition will occur at lower elevations as long as viable seed is available (Rood et al. 1998).

Rate of stage decline. Because willows and cottonwood seedlings are phreatophytic (i.e., their roots must maintain contact with a perennial water source), they are vulnerable to desiccation at steep rates of water table decline. The cohort survival modeling results of J. Stella and others (Stella 2005, Stillwater Sciences 2006) indicate that 50-day rampdown period would be lethal to 50% of a cohort, independent of self-thinning effects, at 0.4 in/day (1 cm/day) for Fremont cottonwood, 0.6 in/day (1.5 cm/day) for narrow-leaved willow, and 1.2 in/day (3 cm/day) for Goodding’s black willow. A given flow ramping rate will produce different stage recession pattern depending on cross-sectional geometry, but most river corridors exhibit dominant channel geometries along large reaches, so some simplifying assumptions may be possible. Spatially-explicit restoration approaches (Stillwater Sciences 2006) are most valuable, because stage-discharge relationships can be modeled independently at each cross section. For example, the SacEFT model (ESSA 2005, 2006) will utilize site-specific stage-discharge relationships for specific cross-sections on the river at the three locations (RM 192, 183, and 172) used in the TNC and CDWR cottonwood studies (Roberts et al. 2002, TNC 2003a, and Morgan and Henderson 2005a, 2005b).

In practice, other human and ecosystem water demands, as well as water infrastructure operating constraints will likely require flow releases that are a hybrid between current river management and an idealized recruitment flow release based on application of a recruitment box model calibrated for local conditions. Some of these tradeoffs will be examined during a later stage of the Sacramento Ecological Flows Study using the TARGETS model of cottonwood seedling initiation that is a component of the SacEFT being developed by ESSA in collaboration with TNC and Stillwater Sciences (ESSA 2005, 2006).

Other hydrograph components

In addition to the elevated flows in spring and a gradual recession rate, several other aspects of the annual hydrograph may be critical to cottonwood and willow recruitment under certain circumstances. These include high winter flow peaks and stable summer baseflows. High-magnitude, short duration winter flows may be necessary prior to a recruitment flow to control vegetation encroachment in the active channel, a common consequence of flow regulation, and to deposit fine sediment on potential seedbed surfaces (sometimes referred to as "encroachment and seedbed preparation flows"[Stillwater Sciences 2003]). Secondly, relatively stable summer and fall baseflows may be required following recruitment to recharge the near-channel water table and ensure that seedlings and young trees have adequate soil moisture to survive the annual dry season ("maintenance flows" [Stillwater Sciences 2003]). This is most critical in drought periods or in particular reaches where the groundwater table has great seasonal fluctuation. The need for these additional hydrograph components will be dependent on local reach conditions and particular climate patterns (e.g., droughts), and should be coordinated with recruitment flow plans.

Interannual flow planning

Environmental conditions during the several years after a recruitment event are critical for survival of young cohorts. Seedlings and saplings face a host of potential mortality agents including desiccation (Rood et al. 1998), scour (McBride and Strahan 1984), prolonged inundation (Kozlowski 1997), water table decline (Stromberg et al. 1996), herbivory (Griggs and Sperber 2003), and disease. Multi-year flow planning should be undertaken to insure that seedlings that recruit one year are not killed by controlled winter releases or large drops in the water table during the dry season. As discussed above (under "Indexing flow planning to water year type"), the frequency of planned recruitment flows should be initially high to mitigate for possible high mortality events due to these factors. Under an adaptive management framework, it is possible that the intervals between recruitment flows might be increased as more information about mortality agents becomes known and the success rate of managed recruitment flows increases.

Variability in the timing and magnitude of recruitment flows is likely to lead to a more diverse and heterogeneous mix of plant species, vegetation types, and habitat structure. We assume that such heterogeneity is closer to natural conditions and more likely to promote a variety of ecological objectives than the simpler riparian communities that would likely develop if recruitment flows always occurred at the same time, peak flow magnitude, and recession rate. Adaptive management could be used to find an appropriate approach to designing and implementing spatially and temporally variable recruitment flows.

Flow planning in the first year or two following a recruitment flow will likely be critical to cohort survival. High flows with the potential to scour young of the year seedlings should be avoided if possible in the winter after a recruitment event. Subsequent summer spring and summer flows should be adequate to maintain soil moisture during the growing season, but not so elevated as to concentrate root growth near the soil surface and increase vulnerability to desiccation during rapid declines later.

Operational issues

Recruitment flow planning may challenge current operational constraints on regulated rivers, especially with regards to flow timing. Currently, managed flood control releases are generally conducted in late winter to maintain flood storage capacity in the reservoirs. Once the snowpack is largely exhausted and the uncertainty in snowmelt timing in the upper watershed reduced, reservoir releases generally decrease in order to store water for summer irrigation needs. These measures have the effect of shifting the bulk of the snowmelt water released earlier in the spring relative to the natural flow regime, and before cottonwood trees begin seed release.

In years with high snowpack volume, reservoirs may conduct sustained releases with flows at a constant high rate well into the summer before returning to baseflow. If the sustained flow outlasts the trees' seed release period, recruitment may be very poor because seedbeds will be submerged while most seeds are washed downstream. If the sustained release ends during the seed release period with a sharp flow rate decline to baseflow levels, any seedlings germinated at high bank elevations will be stranded and desiccated. Though management objectives such as flood control, irrigation and hydropower will largely govern flow operations on managed rivers. However, to the extent that operational flexibility exists in modifying flow timing and ramping rates during spring and summer, the riparian ecosystem stands to gain. Strategically managed, ecologically-sensitive changes in the regulated flow regime may make the difference between a permanently declining pioneer tree population and one that is sustainable.

Likely Benefits for Multiple Target Species

Regulated flow releases for recruitment have a strong potential to benefit other species that are adapted to the large, regular disturbance imposed by the spring snowmelt pulse (Lytle and Poff 2004). For example, winter-run Chinook salmon (*Oncorhynchus tshawytscha*) smolts migrate out of the rivers to the ocean during this event (Section 4.2). The Vernalis Adaptive Management Program on the San Joaquin River is one restoration program that mimics elements of the spring snowmelt pulse to increase outmigrant success (SJRGA 2005). Similarly, managed fall releases to signal upstream migration timing ('attraction flows') are currently implemented every year. These flows may be designed to benefit riparian recruitment if the flow magnitude is sufficient to scour vegetation from potential seedbeds. In these two examples (spring and fall pulse flows), multiple species will benefit from flow restoration measures that are properly coordinated. As currently implemented, VAMP flows occur approximately a month too early to benefit cottonwood and willow species, and the ramping rates are generally too steep to sustain newly-germinated seedlings. Fall attraction flows are currently too modest to scour vegetation or deposit sediment on floodplains.

However, it is also possible that spring recruitment flows could have some negative impacts on other target species. For example, the bank swallow nesting period coincides with the cottonwood recruitment period. Elevated flows in the 22,000 to 37,000 cfs range designed to promote cottonwood recruitment on suitable seedbed elevations 3 to 6 ft (1 to 2 m) above summer baseflow levels (see TNC 2003 for details) might promote some localized episodes of bank erosions that could affect nesting success of some bank swallow pairs (see Chapter 7 for more discussion of this type of effect). Although the risk of significant adverse impacts appears small, monitoring of bank swallow colonies during experimental recruitment flows should be conducted and adaptive management used to refine recruitment flow hydrographs to maximize benefits and minimize negative impacts.

9.8.1.2 Maintenance of lateral migration of the mainstem channel

As discussed in Section 9.2, recruitment of Fremont cottonwood and establishment of riparian forests depends critically on the production of new floodplain and near-river deposits for seedbeds. On the Sacramento River, suitable deposits are generated by progressive migration, (when point bars are deposited on the inside banks of meanders) and by chute cutoff (when abandoned side channels are slowly converted to oxbows and ultimately terrestrialize). As part of any long-term management for improved recruitment of cottonwood on the Sacramento River, it will therefore be necessary to maintain rates of lateral migration and meander bend cutoff that are high enough to produce sufficient quantities of new land along the mainstem. However, the fact that the average rate of production of new land has declined relative to pre-dam conditions, despite a relatively steady average rate of progressive meander migration suggests that the relationship between rates of migration and new land production is complicated, and requires further study. In particular it will be important to determine whether selective enhancement of lateral migration rates leads to local increases in area available for riparian vegetation recruitment.

Even if it doesn't, increases in lateral migration rates will have important benefits for other species. For example, progressive migration is critical for production of bank swallow habitat (Chapter 7), LWD recruitment (Chapter 3), generation of shaded riparian aquatic habitat (Chapters 4–6), scour of pool habitats (Chapter 6), and maintenance of overall habitat complexity within the mainstem. Meander bend cutoff, on the other hand, is the main pathway for generation of off-channel habitats along the Sacramento River (Chapters 3 and 8). As discussed below, lateral channel migration (via progressive migration and meander bend cutoff) can be managed by modulating erodibility of banks and floodplains and by altering the frequency and magnitude of flow, which regulates shear stresses on banks and floodplains surfaces.

Management of flow

As discussed in Chapter 3, the dynamic meandering form of the Sacramento River is maintained by several key components of the flow régime. For example, rates of bank erosion by progressive migration appear to be correlated with cumulative stream power (Figure 3-9), indicating that they are sensitive to the duration of flows above a certain threshold (equal to 29,000 cfs for the reach-specific migration rates shown in Figure 3-9). Over the long term, it will be necessary to verify that flow management for recruitment (see Section 9.8.1.1) supports progressive migration rates that contribute sufficiently abundant fresh deposits for a viable successional process. If management for recruitment or other more natural flow events is insufficient to promote adequate progressive migration, it may be necessary to increase the number and duration of flows above the threshold for bank erosion. This would need to be done in a few select wet years, in the months leading up to the optimal period for recruitment. Enhancement of migration rates during the pre-recruitment interval will likely have the added benefit of contributing to the renewal of bank swallow habitat along the mainstem, so long as they occur prior to the swallow nesting period (see Chapter 7).

Erosion of banks via progressive migration is affected by rates of flow fluctuation, with rapid reductions in stage resulting in elevated pore pressures and increased failure rates of some banks (Chapter 3). While it may be possible to enhance progressive migration via rapid flow fluctuations in select years, this type of flow management may have negative effects on other key habitats and species of the Sacramento River. For example, as discussed in chapter 8, erosion associated with flow fluctuations can have important implications for bank swallows, which require stable banks during their breeding season in early summer. Hence, the timing of any intentional changes in flow-fluctuation rates will need to be carefully considered, along with any other potential effects on important habitats and species.

Chute cutoff frequency is strongly correlated with cumulative overbank flow on the Sacramento River (see Chapter 3, Figure 3-12). This implies that it may be possible to increase rates of chute cutoff by increasing overbank flow during select wet water years. However if the meander bend cutoff frequency is too high, the river may not be able to maintain its sinuosity. There is some indication that cutoff frequency has increased in the post-dam era, and that the sinuosity of meander bend cutoffs has been reduced (Chapter 3). A decrease in sinuosity could, over the long term, reduce the diversity of off- and near-channel habitats along the riparian corridor. It could also promote a negative feedback which inhibits lateral migration; progressive migration rates and the likelihood of meander bend cutoff both increase with increasing sinuosity, so any decreases in sinuosity are likely to lead to decreases in lateral migration rates. The hypothesis that such a negative feedback may eventually develop is corroborated by the fact that the observed increase in cutoff frequency (see Chapter 3) has been accommodated by an increase in partial cutoffs, rather than complete cutoffs, which are generally associated with more rapid lateral migration. Thus, the average rate of lateral migration by cutoffs has decreased as the frequency of cutoff has increased. Moreover, there is some suggestion that the pool of bends that are susceptible to cutoff due to high sinuosity has been depleted by the increased frequency of meander bend cutoff (Micheli and Larsen unpublished). Over time this should eventually lead to a decrease in the frequency of meander bend cutoffs. As both the number of potential cutoffs and the average rate of lateral migration per cutoff decreases, the overall average lateral migration rate of the river is likely to decline. Taken together, the above issues highlight the importance of balancing potential benefits for the riparian corridor as a whole in any decision to increase the frequency of meander bend cutoffs. Key considerations will include:

- how rates of lateral migration are likely to respond over the long term as well as the short term, and
- how reduced habitat complexity might affect the diversity of native plant species and the long-term viability of species such as the western pond turtle (Chapter 8).

It's important to recognize that changes in flow management have the potential to modulate lateral migration rates along the entire migrating length of the river (i.e., from RM 243–143). Moreover, because any changes in flow will affect the river as a whole, it will be difficult to promote increased lateral migration in one area over another. If a more focused approach is desired, with the target being an increase in migration rates at a specific point, it may be possible to achieve the desired effect by modulating erodibility, as discussed next.

Management of erodibility via levee setbacks and removal of riprap

The erodibility of banks is a key regulator of lateral migration rates. As discussed in Chapter 3, the Sacramento River exhibits marked contrasts in meander migration rates depending on geologic conditions which set erodibility of the floodplains. Levees and riprap are artificial constraints that can be removed for immediate increases in local migration rates and area inundated by overbank flows. In the case of levees, if some level of local flood protection is required in a reach that is being considered for restoration, ecosystem benefits can still be achieved if the structures can be set back some distance from the mainstem channel (Golet et al. 2003; Larsen et al. 2006). In some sections of river that have been protected by riprap, it may be possible to achieve benefits by simply discontinuing maintenance of (or “retiring”) the armored banks. Over time these types of management actions should increase sinuosity and the amount of point bar habitat with cross-sectional geometries that are desirable for cottonwood initiation and establishment. As discussed above, a key uncertainty will be whether the rate of new land production increases correspondingly with the increases in progressive migration rates achieved by management of erodibility.

By locally increasing the erodibility of banks via levee setback and riprap removal, it should be possible to promote chute cutoff as well as progressive lateral migration. This would presumably benefit cottonwoods by enhancing establishment along the cutoff recruitment pathway. However as noted in the previous section, enhancement of cutoff can, over the long-term, lead to channel straightening, which may not be desirable many of the Sacramento River focal species and habitat types.

Management that focuses on increasing erodibility can be site specific, unlike management that focuses on changes in flow. Under ideal conditions, sections would be chosen such that increases in erodibility and area connected to the mainstem will lead to direct benefits for key species and habitats. For example, an ideal location for riprap removal would be a bank that is composed of materials that are suitable for bank swallow nesting, and that has a suitable point bar for riparian recruitment on its opposing, inside bend. Ideal locations and designs for levee setbacks can be assessed by running a series of scenarios through a meander migration model (Larsen et al. 2006). The results of such a model would be a key input for incorporation into the SacEFT which help in evaluating the relative benefits of proposed actions in terms of quantifiable metrics such as average annual migration rates and the amount of floodplain reworked. The SacEFT should also help account for how changes in erodibility are likely to interact with any management-related changes in flow releases.

9.8.1.3 Strategic horticultural restoration

There are a number of reasons why horticultural restoration should occur as a complementary conservation strategy to a process-based restoration strategy in many locations. On floodplain surfaces, particularly higher terraces, that have been subject to prior agricultural uses, existing weeds may persist for years on the site and strongly outcompete native species unless active management is undertaken (Peterson 2002). In some cases, particularly lower relative elevation floodplain surfaces that might still be inundated fairly regularly, weed removal and ground clearance may be sufficient to jump start natural recruitment processes. In other cases, especially on infrequently inundated sites, active revegetation using horticultural techniques may be the only means for re-establishing native plant communities within a timeframe that is acceptable to land managers (Whisenant 1999).

A recent example of the apparent success of “provide habitat and they will come” approach in the Central Valley comes from the return of the Least Bell's Vireo to a restoration site in the San Joaquin Basin. There was a confirmed nesting of Least Bell's Vireo in June 2005 at San Joaquin River National Wildlife Refuge, and again in 2006, on a CALFED habitat restoration site that was initiated in 2002. This is first confirmed nesting of this species in the Central valley since 1919 (despite exhaustive searches for the species in the 1970s and 1980s). Prior to these recent records, the Least Bell's Vireo has not been seen or heard in the last 60 years. It was considered extirpated from Central Valley and federally listed as endangered in 1986. This apparent success story suggests that the adaptive management approach to habitat restoration can be successful. In this case, a shrubby understory was planted to create a varied pattern of vegetation mimicking natural floodplains on the assumption that this was a key component of suitable breeding habitat for the Least Bell's Vireo and other bird species. A similar success story of a target bird species using restoration sites for nesting and foraging is occurring on the Sacramento River with the recent records of yellow-billed cuckoos using restoration sites (Small et al. 2000, TNC unpublished data).

9.8.1.4 Prioritized actions to eradicate and control invasive plant species

A goal of reducing the acreage and distribution of invasive, non-native species and vegetation types should be an important part of river corridor restoration and management efforts. Develop a control or eradication strategy for each of the primary non-native invasive species to ensure that restoration actions do not inadvertently promote the spread of these species, and that restoration actions increase control of these species as much as possible.

The following actions should be considered during restoration plan development and implementation.

- Eradicate isolated occurrences of invasive non-native plants. Eradicating non-native plant species is difficult and usually unattainable. Complete eradication is, however, a potentially feasible goal where non-native species occur as small, isolated patches. Eradicating these types of species would likely require an integrated pest management approach (e.g., a combination of physical removal and limited herbicide application) to remove the existing stands, monitoring of sites to identify any resprouting of treated stands, maintenance to treat any resprouting, and river-wide monitoring to identify any other occurrences or recent introductions of the species (see below).
- Minimize the introduction of non-native plant species and other native species, such as Argentine ants in planted stock, when implementing restoration actions. Because many non-native species can out-compete native species in colonizing disturbed areas, non-native species can interfere with the success of restoration actions, particularly when restoration actions (such as dispersal flows, floodplain grading, or channel modifications) create opportunities for the dispersal and establishment of the invasive species. The biology of potential invasive species and the techniques available to control their spread should therefore be considered when developing restoration strategies and actions.
- Promote processes and conditions that encourage native plant species recruitment over non-native species. Habitat fragmentation, alteration of historical disturbance regimes (such as flooding and fire), and increased nutrient delivery by adjacent land uses are just a few of the ways humans have altered riparian areas such that non-native plant species have a competitive advantage over natives. Conserving and expanding existing native habitat patches will not only reduce edge habitat (which is more easily colonized by non-native species), but will also provide necessary sources for native seed dispersal. Restoring natural fluvial processes to the extent possible will provide the conditions necessary to recruit native riparian species (such as bare, moist seedbeds and thinning of the understory), while scouring and inhibiting non-native species. Actions to improve water quality will also help improve conditions for natives that are sensitive to elevated levels of nutrients and other pollutants.
- Re-establish native plants in areas where non-native species are removed or treated. Removal of invasive species is not guaranteed to remove the invasive impacts. Locally extirpated native species may require re-introduction to the site.
- Establish a river-wide monitoring program. Frequent monitoring of the river corridor will be needed to identify recent introductions and infestations. Once a species has become widespread and abundant, mechanical and/or chemical removal can be prohibitively expensive, and even after an invasive species is removed, it frequently re-invades, requiring ongoing treatment. Regular monitoring of the river corridor for new

introductions or resprouting of treated stands will help identify small, isolated patches of invasive non-native plants that can be more feasibly eradicated before they become widespread.

9.8.2 Potential performance metrics

- Total area and relative abundance of cottonwood forest , mixed riparian forest, and valley oak riparian forest
- Rate of riparian forest creation (average area added per year) through natural processes (channel meander migration or cutoffs) or human factors (horticultural restoration)
- Rate of riparian forest loss (average area lost per year) due to natural causes (e.g., bank erosion) and human factors (land clearance for agriculture, bank revetment, etc.)
- Frequency of successful Fremont cottonwood recruitment events (number per decade)
- Abundance of non-native invasive plant species (e.g., area dominated by giant reed or Himalaya berry)

9.8.3 Key hypotheses and uncertainties

- Uncertainty about how often new cohorts of cottonwoods and willows should be recruited
    - Is a frequency of successful cottonwood recruitment once every 5 or 10 years on average sufficient to develop a dynamic and complex mosaic of vegetation and habitat types that will support a diverse assemblage of native plants and wildlife, including riparian songbirds?
    - How much should the timing and magnitude of recruitment flows vary to promote more heterogeneous mixes of cottonwoods and willows and topographic patterns of establishment?
- Uncertainty about riparian vegetation successional processes
    - What is the relative importance of point bar vs. oxbow/meander cutoff pathways for recruitment of riparian vegetation among the various reaches? Greco's dissertation work (1999) indicated that the channel cutoff pathway may account for one-third or more of the cottonwood and mixed riparian forest stands that established between Woodson Bridge and Chico Landing during his period of study. Similar studies are needed to determine if this pathway is of similar or greater importance in other reaches, particularly from Red Bluff to Woodson Bridge.
    - Has the relative importance of each pathway changed from historical conditions? If so, what are the implications for the long-term trajectory of cottonwood recruitment?
    - What other factors affect succession and transition probabilities? Is there a risk of shifting to an alternative stable state that would be less desirable based on human restoration goals?
    - What affects diversity and abundance of native understory species? How can we manage for native diversity and reduced invisibility to non-native species? Research currently underway by CSU Chico and UC Santa Cruz researches should help improve our understanding of these issues.
    - How do other ecosystem conditions and processes change during succession (nutrient cycling, soil development, etc.)?
- Uncertainty regarding hypothesis that plant and bird assemblages respond to different types of habitat revegetation (e.g., passive restoration vs. horticultural restoration; passive revegetation via point bar vs oxbow/meander cutoff pathways)?

  - Which overstory and understory plant species are most important for nesting, foraging, cover and roosting? Continuing existing bird monitoring and habitat relationship studies should greatly improve our understanding of these issues.
- Uncertainty about which riparian songbird species require key habitat elements (e.g., sloughs, oxbows, grasslands or herbaceous vegetation types) in addition to stands of native riparian forest.
  - How important are habitat elements such as sloughs or oxbow lakes to reproductive success of native songbird species? How does variation in the density and distribution of such features across the landscape affect habitat suitability and reproductive success for different songbirds?
- Uncertainty about how much nest predators and parasites affect native riparian songbird reproductive success, and how well their effects can be controlled solely via habitat restoration.
  - Is active management of brown-headed cowbird or non-native nest predator populations required to restore productive and sustainable populations of certain native riparian songbirds?

9.9 References

Alexander, C. 2004. Riparian Initiation, Scour and Chinook Egg Survival Models for the Trinity River. Notes from a Model Review Meeting held September 3rd - 5th, 2003. Draft report prepared by ESSA Technologies Ltd., Vancouver, BC for McBain and Trush, Arcata, CA. 29 pp.

Amlin, N. M., and S. B. Rood. 2002. Comparative tolerances of riparian willows and cottonwoods to water-table decline. Wetlands 22: 348–346.

Andelman, S. J. and W. F. Fagan. 2000. Umbrellas and flagships: efficient conservation surrogates or expensive mistakes? Proceedings of the National Academy of Sciences 97: 5954-5959.

Anderson, B., R. Banks, S. Laymon, and R. McKernan. 1994. Western yellow-billed cuckoo. Pages 192-193 *in* C. G. Thelander and M. Crabtree, editors. Life on the edge: a guide to California's endangered natural resources: wildlife. Biosystems Books, Santa Cruz, California.

Asplund, K. K., and M. T. Gooch. 1988. Geomorphology and the distributional ecology of Fremont cottonwood (*Populus fremontii*) in a desert riparian canyon. Desert Plants 9: 17–27.

Bair, J. H. 2001. Riparian woody plant initiation, establishment, and mortality on rehabilitated banks of the Trinity River, California. Master's thesis. Humboldt State University, Arcata, California.

Baker, W. L. 1990. Climatic and hydrologic effects on the regeneration of Populus angustifolia James along the Animas River, Colorado. Journal of Biogeography 17: 59–73.

Baker, W. L, and G. M. Walford. 1995. Multiple stable states and models of riparian vegetation succession on the Animas River, Colorado. Annals of the Association of American Geographers 85: 320-338.

Battles, J., J. Stella, B. Orr, and M. Scott. 2005. Restoring non-equilibrium riparian communitites in disturbance-altered ecosystems: implications for river management and climate change. Proposal submitted to CALFED Science Program.

The Bay Institute. 1998. From the Sierra to the sea: the ecological history of the San Francisco Bay-Delta watershed. San Rafael, California.

Bendix, J. 1994. Among-site variation in riparian vegetation of the southern California Transverse Ranges. The American Midland Naturalist 132: 136–151.

Bendix, J. 1998. Impact of a flood on southern California riparian vegetation. Physical Geography 19: 162-174.

Bendix, J. 1999. Stream power influence on southern California riparian vegetation. Journal of Vegetation Science 10: 243–252.

Bendix, J., and C. R. Hupp. 2000. Hydrological and geomorphological impacts on riparian plant communities. Hydrological Processes 14: 2977-2990.

Bonn, A., K. Hagen, and D. Wohlgemuth-von Reiche. 2002. The significance of flood regimes for carabid beetle and spider communities in riparian habitats - a comparison of three major rivers in Germany. River Research and Applications 18: 43–64.

Braatne, J. H., S. B. Rood, and P. E. Heilman. 1996. Life history, ecology, and conservation of riparian cottonwoods in North America. Pages 57-86 *in* R. F. Stettler, H. D. Bradshaw Jr., P. E. Heilman and T. M. Hinckley, editors. Biology of *Populus* and its implication for management and conservation. NRC Research Press, National Research Council of Canada, Ottawa.

Bradley, C. E., and D. G. Smith. 1986. Plains cottonwood recruitment and survival on a prairie meandering river floodplain, Milk River, southern Alberta and northern Montana. Canadian Journal of Botany 64: 1433–1442.

Buer, K. 1994. Sacramento River bank erosion investigation memorandum progress report. Internal memorandum to R. Scott and L. Brown from K. Buer, Chief, Geology Section, California Department of Water Resources, Northern District, Red Bluff.

Buer, K., D. Forwalter, M. Kissel, and B. Stohler. 1989. The middle Sacramento River: human impacts on physical and ecological processes along a meandering river. Pages 22-32 *in* D. L. Abell, editor. Proceedings of the California riparian systems conference: protection, management, and restoration for the 1990s. General Technical Report PSW-110. USDA Forest Service, Pacific Southwest Forest and Range Experiment Station, Berkeley, California.

California Resources Agency. 1998. Sacramento River Conservation Area handbook. Draft report. Prepared by Sacramento River Advisory Council for The Resources Agency of California under the SB 1086 Program. California Department of Water Resources, Sacramento.

CDFG (California Department of Fish and Game). 1983. Sacramento River and tributaries bank protection and erosion control investigation--evaluation of impacts on fisheries. Final Report. CDFG, Bay-Delta Fishery Project.

CDWR (California Department of Water Resources). 2001. Indicators of Hydrologic Alteration data files on CD-ROM. Memorandum to Glenn Pearson from Stacy Cepello and Shawn Pike, CDWR, Sacramento, California. 23 February.

CDWR. 2005. California Water Plan Update 2005: A Framework for Action. Bulletin 160-05, California Department of Water Resources, Sacramento, CA.

Collinge, S. K., M. Holyoak, C. B. Barr, and J. T. Marty. 2001. Riparian habitat fragmentation and population persistence of the threatened valley elderberry longhorn beetle in central California. Biological Conservation 100: 103–113.

Cooper, D. T. 1990. Populus deltoides Bartr. Ex Marsh. var. deltoides: Eastern cottonwood. Pages 530–535 *in* R. M. Burns and B. H. Honkala, editors. Silvics of North America: hardwoods, Vol. 2. U.S. Department of Agriculture, Agriculture Handbook 654.

Cordes, L. D., F. M. R. Hughes, and M. Getty. 1997. Factors affecting the regeneration and distribution of riparian woodlands long a northern prairie river: the Red Deer River, Alberta, Canada. Journal of Biogeography 24: 675–695.

Cordes, L. D. 1991. The distribution and age structure of cottonwood stands along the Lower Bow River, Alberta. Pages 13–24 *in* S. B. Rood and J. M. Mahoney, editors. Proceedings of the biology and management of southern Alberta's cottonwoods conference.

Cox, G. 1999. Alien species in North America and Hawaii: impacts on natural ecosystems. Island Press, Washington, D. C.

CPIF and RHJV (California Partners in Flight and Riparian Habitat Joint Venture). 1998. The draft riparian bird conservation plan: a strategy for reversing the decline of birds and associated riparian species in California.

DiTomaso, J. 2006. New weed alerts! CAL-IPC News Winter, 13: 4-5.

Dobkin, D. S. 1994. Conservation and management of neotropical migrant land birds in the northern Rockies and Great Plains. University of Idaho Press, Moscow, Idaho.

Eckenwalder, J. E. 1996. Systematics and evolution of *Populus* Pages 7-32 *in* R. F. Stettler, H. D. Bradshaw Jr., P. E. Heilman and T. M. Hinckley, editors. Biology of *Populus* and its implication for management and conservation. NRC Research Press, National Research Council of Canada, Ottawa.

ESSA Technologies Ltd. 2005. Sacramento River Decision Analysis Tool: workshop backgrounder. Prepared by ESSA Technologies Ltd., Vancouver, British Columbia for The Nature Conservancy, Chico, California.

ESSA Technologies Ltd. 2006. Sacramento River Ecological Flows Tool (SacEFT): design guidelines. Prepared for by ESSA Technologies Ltd., Vancouver, British Columbia for The Nature Conservancy, Chico, California.

Fremier, A. K. 2003. Floodplain age modeling techniques to analyze channel migration and vegetation patch dynamics on the Sacramento River, California. Master's thesis. University of California, Davis.

Friedman, J. M., M. L. Scott, and G. T. Auble. 1997. Water management and cottonwood forest dynamics along prairie streams. Pages 49-71 *in* F. L. Knopf and F. B. Samson, editors. Ecology and conservation of Great Plains vertebrates. Springer-Verlag, New York.

Friedman, J. M., W. R. Osterkamp, M. L. Scott, and G. T. Auble. 1998. Downstream effects of dams on channel geometry and bottomland vegetation: regional patterns in the Great Plains. Wetlands 18: 619–633.

Gaines, D. 1974. Review of the status of the yellow-billed cuckoo in California: Sacramento Valley populations. Condor 76: 204–209.

Gaines, D. 1977. Current status and habitat requirements of the yellow-billed cuckoo in California. California Department of Fish and Game, Sacramento.

Gardali, T., S. L. Small, N. Nur, G. R. Geupel, G. Ballard, and A. L. Holmes. 2005. Monitoring songbirds in the Sacramento Valley (1993-2003): population health, management information, and restoration evaluation. PRBO Contribution No. 1233 Prepared by Point Reyes Bird Observatory, Stinson Beach, California for The Nature Conservancy, U.S. Fish and Wildlife Service, and River Partners.

Gardali, T., A. L. Holmes, S. L. Small, N. Nur, G. R. Geupel, and G. H. Golet. 2006. Abundance patterns of landbirds in restored and remnant riparian forests on the Sacramento River, California, U.S.A. Restoration Ecology 14: 391-403.

Golet, G. H., D. L. Brown, E. E. Crone, G. R. Geupel, S. E. Greco, K. D. Holl, D. E. Jukkola, G. M. Kondolf, E. W. Larsen, F. K. Ligon, R. A. Luster, M. P. Marchetti, N. Nur, B. K. Orr, D. R. Peterson, M. E. Power, W. E. Rainey, M. D. Roberts, J. G. Silveira, S. L. Small, J. C. Vick, D. S. Wilson, and D. M. Wood. 2003. Using science to evaluate restoration efforts and ecosystem health on the Sacramento River Project, California. Pages 368–385 *in* P. M. Faber, editor. California riparian systems: processes and floodplain management, ecology, and restoration. 2001 Riparian habitat and floodplain conference proceedings. Riparian Habitat Joint Venture, Sacramento, California.

Greco, S. E. 1999. Monitoring riparian landscape change and modeling habitat dynamics of the yellow-billed cuckoo on the Sacramento River, California. Doctoral dissertation. University of California, Davis.

Greco, S. E., A. K. Fremier, E. W. Larsen, and R. E. Plant. In press. Tracking surficial patterns of floodplain chronology on a large meandering river: analysis of land production rates and riparian vegetation distribution over land age gradients.

Greco, S. E., E. H. Girvetz, E. W. Larsen, J. P. Mann, J. L. Tuil, and C. Lowney. In review (2006). A method to model a relative elevation topographic surface of a large alluvial river floodplain and riparian ecological application. Submitted to Landscape Research.

Gregory, S. V., F. J. Swanson, W. A. McKee, and K. W. Cummins. 1991. An ecosystem perspective of riparian zones. BioScience 41: 540–551.

Griffin, J. R., and W. B. Critchfield. 1972. The distribution of forest trees in California. USDA Forest Service, Berkeley, California.

Griggs, F. T., and T. Sperber. 2003. The interaction of biological and physical processes and the long-term survival of riparian forests at the San Joaquin National Wildlife Refuge. Second Biennial CALFED Science Conference. January 14-16, 2003. Sacramento, CA.

Griggs, T., and S. Small. 2000. Riparian vegetation white paper for CALFED. Draft of 27 March 2000.

Groffman, P. M., N. L. Law, K. T. Belt, L. E. Band, and G. T. Fisher. 2004. Nitrogen fluxes and retention in urban watershed ecosystems. Ecosystems 7: doi:10.1007/s10021-003-0039-x.

Guilloy-Froget, H., E. Muller, N. Barsoum, and F. M. R. Hughes. 2002. Dispersal, germination, and survival of *Populus nigra* L. (Salicaceae) in changing hydrologic conditions. Wetlands 22: 478–488.

Halterman, M. D. 1991. Distribution and habitat use of the yellow-billed cuckoo (*Coccyzus americanus occidentalis*) on the Sacramento River, California, 1987–1990. Master's thesis. California State University, Chico.

Harvey, M. D. 1989. Meanderbelt dynamics of the Sacramento River, California. Pages 54–59 in D. L. Abell, editor. Proceedings of the California riparian systems conference: protection, management, and restoration for the 1990s. General Technical Report PSW-110. USDA Forest Service, Pacific Southwest Forest and Range Experiment Station, Berkeley, California.

Hickman, J. C., editor. 1993. The Jepson manual: Higher plants of California. University of California Press, Berkeley.

Holl, K. D., and E. E. Crone. 2004. Applicability of landscape and island biogeography theory to restoration of riparian understorey plants. Journal of Applied Ecology 41: 922–933.

Horton, J. S., F. C. Mounts, and J. M. Draft. 1960. Seed germination and seedling establishment for phreatophyte species. Research Paper No 48. USDA Forest Service, Rocky Mountain Forest and Range Experimental Station.

Howe, W. H., and F. L. Knopf. 1991. On the imminent decline of Rio Grande cottonwoods in central New Mexico. The Southwestern Naturalist 36: 218–224.

Hughes, F. M. R. 1994. Environmental change, disturbance and regeneration in semi-arid floodplain forests. Pages 321–345 *in* A. C. Millington and K. Pye, editors. Environmental change in drylands: biogeographical and geomorphological perspectives. John Wiley & Sons, New York.

Hupp, C. R. 1983. Vegetation pattern on channel features in the Passage Creek Gorge, Virginia. Castanea 48: 62-72.

Hunt, J. W. 2004. Comparison of epigeal beetle assemblages in remnant and restored riparian forests on the middle Sacamento River, California. Master's thesis. California State University, Chico.

Hunter, W. C., B. S. Anderson, and R. D. Ohmart. 1987. Avian community structure changes in a mature floodplain forest after extensive flooding. Journal of Wildlife Management 51: 495–502.

Huxell, G. R. 2000. The effect of the Argentine ant on the threatened valley elderberry longhorn beetle. Biological Invasions 2: 81–85.

Jamieson, B., and J. Braatne. 2001. Riparian cottonwood ecosystems and regulated flows in Kootenai and Yakima sub-basins: impacts of flow regulation on riparian cottonwood forests of the Yakima River. Technical Report 2000-2001. BPA Report No. DOE/BP-00000005-3. Bonneville Power Administration, Portland, Oregon.

Jenny, H. 1941. The factors of soil formation. McGraw-Hill, New York, New York.

Johansson, M. E., C. Nilsson, and E. Nilsson. 1996. Do rivers function as corridors for plant dispersal? Journal of Vegetation Science 7: 593–598.

Johnson, R. R., and L. T. Haight. 1984. Riparian problems and initiatives in the American Southwest: a regional perspective. Pages 404–412 in R. E. Warner and K. M. Hendrix, editors California riparian systems: ecology, conservation, and productive management. University of California Press, Berkeley.

Johnson, W. C. 2000. Tree recruitment and survival in rivers; influence of hydrological processes. Hyrdological Processes 14: 3051–3074.

Johnson, W. C., R. L. Burgess, and W. R. Keammerer. 1976. Forest overstory vegetation and environment on the Missouri River floodplain in North Dakota. Ecological Monographs 46: 59–84.

Katibah, E. F. 1984. A brief history of riparian forests in the Central Valley of California. Pages 23-29 *in* R. E. Warner and K. M. Hendrix, editors. California riparian systems: ecology, conservation, and productive management. University of California Press, Berkeley.

Knopf, F. L., J. A. Sedgwick, and R. W. Cannon. 1988. Guild structure of a riparian avifauna relative to seasonal cattle grazing. Journal of Wildlife Management 52: 280–290.

Kondolf, G. M., and P. R. Wilcock. 1996. The flushing flow problem: defining and evaluating objectives. Water Resources Research 32: 2589–2599.

Kondolf, G. M., T. Griggs, E. W.Larsen, S. McBain, M. Tompkins, J. G. Williams, and J. Vick. 2000. Flow regime requirements for habitat restoration along the Sacramento River between Colusa and Red Bluff. CALFED Bay Delta Program, Integrated Storage Investigation, Sacramento, California.

Kozlowski, T. T. 1997. Responses of woody plants to flooding and salinity. Pages 1-29 *in* Tree Physiology Monograph No. 1. Heron Publishing, Victoria, Canada.

Landres, P. B., J. Verner, J. W. Thomas. 1988. Ecological uses of vertebrate indicator species: a critique. Conservation Biology 2: 316-328.

Larsen, E. W., E. H. Girvetz, and A. K. Fremier. 2006. Assessing the effects of alternative setback channel constraint scenarios employing a river meander migration model. Environmental Management 37: 880-897. doi:10.1007/s00267-004-0220-9.

Laymon, S. A. 1998. Yellow-billed cuckoo (*Coccycus americanus*). CPIF Riparian Bird Conservation Plan species account. California Partners in Flight (CPIF), Point Reyes Bird Observatory, Stinson Beach, California.

Laymon, S. A., and M. D. Halterman. 1989. A proposed habitat management plan for yellow-billed cuckoos in California. Pages 272–277 *in* D. L. Abell, editor. Proceedings of the California riparian systems conference: protection, management, and restoration for the 1990s. General Technical Report PSW-110. USDA Forest Service, Pacific Southwest Forest and Range Experiment Station, Berkeley, California.

Laymon, S. A., P. L. Williams, and M. D. Halterman. 1997. Breeding status of the yellow-billed cuckoo in the South Fork Kern River Valley, Kern County, California: summary report 1985–

1996. Administrative Report. USDA Forest Service, Sequoia National Forest, Cannell Meadow Ranger District.

Ligon, F. K., W. E. Dietrich, and W. J. Trush. 1995. Downstream ecological effects of dams: a geomorphic perspective. BioScience 45: 183–192.

Likens, G. E., F. H. Bormann, N. M. Johnson, D. W. Fisher, and R. S. Pierce. 1970. Effects of forest cutting and herbicide treatment on nutrient budgets in the Hubbard Brook watershed-ecosystem. Ecological Monographs 40: 23–47.

Lytle, D. A., and D. M. Merritt. 2004. Hydrologic regimes and riparian forests: a structured population model for cottonwood.

Lytle, D. A., and N. L. Poff. 2004. Adaptation to natural flow regimes. Trends in Ecology & Evolution 19: 94–100.

Mahoney, J. M., and S. B. Rood. 1991. A device for studying the influence of declining water table on poplar growth and survival. Tree Physiology 8: 305–314.

Mahoney, J. M., and S. B. Rood. 1992. Response of a hybrid poplar to water-table decline in different substrates. Forest Ecology and Management 54: 141–156.

Mahoney, J. M., and S. B. Rood. 1993. A model for assessing the effects of altered river flows on the recruitment of riparian cottonwoods. B. Tellman, H. J. Cortner, M. G. Wallace, L. F. DeBano and R. H. Hamre, editors. Riparian management: common threads and shared interests. General Technical Report RM-226. USDA Forest Service.

Mahoney, J. M., and S. B. Rood. 1998. Streamflow requirements for cottonwood seedling recruitment - an integrative model. Wetlands 18: 634–645.

Malanson, G. P. 1993. Riparian landscapes. Cambridge University Press, Cambridge, England.

Malanson, G. P., and D. R. Butler. 1991. Floristic variation among gravel bars in a subalpine river. Arctic and Alpine Research 23: 273–278.

McBain and Trush. 2000. Habitat restoration plan for the lower Tuolumne River corridor. Prepared for Tuolumne River Technical Advisory Committee (TRTAC) by McBain and Trush, Arcata, with assistance from U.S. Fish and Wildlife Service Anadromous Fish Restoration Program (AFRP).

McBain and Trush. 2002. San Joaquin River restoration study background report. Prepared for Friant Water Users Authority, Lindsay, California and Natural Resources Defense Council, San Francisco, California by McBain and Trush, Arcata.

McBride, J. R., and J. Strahan. 1984. Fluvial processes and woodland succession along Dry Creek, Sonoma County, California. Pages 110–119 *in* R. E. Warner and K. M. Hendrix, editors. California riparian systems: ecology, conservation, and productive management. University of California Press, Berkeley.

McBride, J. R., N. Sugihara, and E. Norberg. 1989. Growth and survival of three riparian woodland species in relation to simulated water table dynamics. Environment, Health, and Safety Report No. 009.4-89.3. Prepared by University of California, Department of Forestry and Resource Management, Berkeley for Pacific Gas and Electric Company, Department of Research and Development, San Ramon, California.

Micheli, E. R., and E. W. Larsen. In preparation. *N/A River channel cutoff dynamics, Sacramento River, California, USA

Mitsch, W. J., and J. G. Gosselink. 1993. Wetlands. Second edition. Van Nostrand Reinhold, New York.

Morgan, T. L. 2005. Hydrological and physiological factors controlling Fremont cottonwood seedling establishment along the Sacramento River, California: surface and alluvial groundwater relations, ecophysiological analysis from 2002-2004 field seasons, and cottonwood water use determination using stable isotope analysis. Master's thesis. California State University, Chico.

Morgan, T. L., and A. Henderson. 2005a. Field observations of cottonwood seedling survival at River Mile 192.5 during 2002 and 2003, Sacramento River, California. 30 March. Memorandum report. Prepared for California Department of Water Resources, Northern District.

Morgan, T. L., and A. Henderson. 2005b. Cottonwood seedling monitoring during 2004 and 2005 along the Sacramento River, California. 30 December. Memorandum draft report. Prepared for California Department of Water Resources, Northern District.

Morrison, M. L., K. S. Smallwood, and L. S. Hall. 2003. Creating habitat through plant relocations: lessons from valley elderberry longhorn beetle mitigation. Ecological Restoration 21: 95–100.

Naiman, R. J., and H. Decamps. 1997. The ecology of interfaces: riparian zones. Annual Review of Ecology and Systematics 28: 621–658.

Naiman, R. J., H. Decamps, and M. E. McClain. 2005. Riparia: ecology, conservation, and management of streamside communities. Elsevier Academic Press, San Diego, California.

NRC (National Research Council). 2002. Riparian areas: functions and strategies for management. National Academy Press, Washington, D. C.

Nur, N., G. Ballard, and G. R. Geupel. 2005. The response of riparian bird species to vegetation and local habitat features in the Central Valley, California: a multi-species approach across spatial scales. Pages. 40–83 *in* Gardali, T., S. L. Small, N. Nur, G. R.Geupel, G. Ballard, and A. L. Holmes, editors. Monitoring songbirds in the Sacramento Valley (1993-2003): population health, management information, and restoration evaluation. PRBO unpublished report, Contribution # 1233.

Oliver, C. D., and B. L. Larson. 1996. Forest stand dynamics: update edition. Update edition. John Wiley and Sons, Inc.

Pelzman, R. J. 1973. Causes and possible prevention of riparian plant encroachment on anadromous fish habitat. Environmental Services Branch Administrative Report 73-1. California Department of Fish and Game, Region 1.

Peterson, D. 2002. The development of an alternative restoration strategy for Sacramento River riparian forests. Masters Thesis. California State University, Chico.

Pickett, S. T. A., and J.N. Thompson. 1978. Patch dynamics and the design of nature reserves. Biological Conservation 13: 27–37.

Poff, N. L., J. D. Allan, M. B. Bain, J. R Karr, K. R Prestegaard,. B. D. Richter, and J. C. Stromberg. 1997. The natural flow regime: a paradigm for river conservation and restoration. BioScience 47: 521–529.

Power, M. E. and W. E. Rainey. 2000. Food webs and resource sheds: towards spatially delimiting trophic interactions. Pages 291-314 *in* M. J. Hutchings, E. A.John and A. J. A. Stewart, editors. Ecological Consequences of Habitat Heterogeneity. Blackwell Scientific, Oxford, UK.

Randall, J., and M. Hoshovsky. 2000. California's wildland invasive plants. C. Brossard, J. C. Randall and M. Hoshovsky, editors. University of California Press, Berkeley.

Reichenbacher, F. W. 1984. Ecology and evolution of southwestern riparian plant communities. Desert Plants 6: 15–23.

Reid, D. E. 1991. Cottonwoods on the Milk River. Pages 35-42 *in* S. B. Rood and J. M. Mahoney, editors. Proceedings of the biology and management of southern Alberta's cottonwoods conference.

RHJV (Riparian Habitat Joint Venture). 2000. The riparian bird conservation plan: a strategy for reversing the decline of riparian associated birds in California. Version 1.0. California Partners in Flight, Point Reyes Bird Observatory, Stinson Beach, California. http://www.prbo.org/CPIF/Consplan.html.

RHJV (Riparian Habitat Joint Venture). 2004. The riparian bird conservation plan: a strategy for reversing the decline of riparian associated birds in California. Version 2.0. California Partners in Flight, Point Reyes Bird Observatory, Stinson Beach, California. http://www.prbo.org/calpif/riparian.v-2.pdf

River Partners. 2004. Riparian habitat restoration: Sacramento River Wildlife Area, Moulton Weir Unit, Colusa County, California. Final report. Prepared by River Partners, Chico, California for California Department of Fish and Game, Wildlife Conservation Board.

Roberts, M. D., D. R. Peterson, D. E. Jukkola, and V. L. Snowden. 2002. A pilot investigation of cottonwood recruitment on the Sacramento River. The Nature Conservancy, Sacramento River Project, Chico, California.

Rood, S. B., C. R. Gourley, E. M. Ammon, L. G. Heki, J. R. Klotz, M. L. Morrison, D. Mosley, G. G. Scoppettone, S. Swanson, and P. L. Wagner. 2003. Flows for floodplain forests: a successful riparian restoration. BioScience 53: 647–656.

Rood, S. B., A. R. Kalischuk, T. Bond, L. Bridges, R. Cain, P. Fay, M. Heller, K. Larter, K. Mikkelborg, G. Neufeld, M. L. Polzin, C. Rogers, J. Schmaltz, C. Vair, and L. Weaver. 1997. Canyonlands cottonwoods: mortality of Fremont cottonwoods in the Matheson Wetlands Preserve along the Colorado River at Moab, Utah. The Nature Conservancy.

Rood, S. B., A. R. Kalischuk, and J. M. Mahoney. 1998. Initial cottonwood seedling recruitment following the flood of the century of the Oldman River, Alberta, Canada. Wetlands 18: 557–570.

Rood, S. B., and J. M. Mahoney. 1990. Collapse of riparian poplar forests downstream from dams in western prairies: probable causes and prospects for mitigation. Environmental Management 14: 451–464.

Rood, S. B., and J. M. Mahoney. 2000. Revised instream flow regualtion enables cottonwood recruitment along the St. Mary River, Alberta, Canada. Rivers 7: 109–125.

Rood, S. B., G. M. Samuelson, J. H. Braatne, C. R. Gourley, F. M. R. Hughes, and J. M. Mahoney. 2005. Managing river flows to restore floodplain forests. Frontiers in Ecology and the Environment 3: 193–201.

Saab, V. A., C. E. Bock, T. D. Rich, and D. S. Dobkin. 1995. Livestock grazing effects on migratory landbirds in western North America. Pages 311–353 *in* T. E. Martin and D. M. Finch, editors. Ecology and management of neotropical migratory birds: a synthesis and review of critical issues. Oxford University Press, New York.

Sawyer, J. and T. Keeler-Wolf. In press. Manual of California Vegetation. Second edition. California Native Plant Society, Sacramento.

Scott, M., G. T. Auble, and J. M. Friedman. 1997. Flood dependency of cottonwood establishment along the Missouri River, Montana, USA. Ecological Applications 7: 677–690.

Scott, M. L., J. M. Friedman, and G. T. Auble. 1996. Fluvial processes and the establishment of bottomland trees. Geomorphology 14: 327–339.

Segelquist, C. A., M. L. Scott, and G. T. Auble. 1993. Establishment of *Populus deltoides* under simulated alluvial groundwater decline. The American Midland Naturalist 130: 274–285.

Shafroth, P. B., J. C. Stromberg, and D. T. Patten. 2002. Riparian vegetation response to altered disturbance and stress regimes. Ecological Applications 12: 107-123.

Shanafield, A. N. 1983. Alder, cottonwood, and sycamore distribution and regeneration along the Nacimiento River, California. Pages 196–202 *in* R. E. Warner and K. M. Hendrix, editors. California riparian systems. University of California, Davis, California.

Sigafoos, R. S. 1964. Botanical evidence of floods and flood-plain deposition. Geological Survey Professional Paper 485-A. U.S. Geological Survey, Washington, D. C.

Simberloff, D. 1998. Flagships, umbrellas and keystones: is single species management passé in the landscape era? Biological Conservation 83: 247-257.

SJRGA (San Joaquin River Group Authority). 2005. 2004 Annual technical report on the implementation and monitoring of the San Joaquin River Agreement and the Vernalis Adaptive Management Plan.

Small, S. S., J. DeStaebler, G. R. Geupel, and A. King. 1999. Landbird response to riparian restoration on the Sacramento River system: preliminary results of the 1997 and 1998 field season. Prepared for The Nature Conservancy and U.S. Fish and Wildlife Service by Point Reyes Bird Observatory, Stinson Beach, California.

SRCAF (Sacramento River Conservation Area Forum). 2003. Sacramento River Conservation Area Forum handbook. Prepared for The Resources Agency, State of California by the Sacramento River Advisory Council; revised and updated by the Sacramento River Conservation Area Forum, Red Bluff, California. http://www.sacramentoriver.ca.gov.

Stella, J. C., J. J. Battles, B. K. Orr, and J. R. McBride. In press. Synchrony of seed dispersal, hydrology and local climate in a semi-arid river reach in California. Ecosystems.

Stella, J. C., J. J. Battles, J. R. McBride, B. K. Orr. In review. Survival, growth, and physiology of riparian tree seedlings in response to simulated river stage decline. Submitted to Tree Physiology.

Stella, J. C. 2005. A field-calibrated model of pioneer riparian tree recruitment for the San Joaquin Basin, CA. University of California, Berkeley, Berkeley, CA.

Stella, J., J. Vick, and B.K. Orr. 2003. Riparian vegetation dynamics on the Merced River. Pages 302-314 *in* P. M. Faber, editor. California riparian systems: processes and floodplain management, ecology, and restoration. 2001 Riparian habitat and floodplain conference proceedings. Riparian Habitat Joint Venture, Sacramento, California.

Stillwater Sciences. 2001. Merced Restoration Plan Phase II. Volume II: baseline evaluations; geomorphic and riparian vegetation investigations. Prepared by Stillwater Sciences, Berkeley, California for CALFED, Sacramento, California.

Stillwater Sciences. 2003. Draft restoration strategies for the San Joaquin River: adaptive management report. Prepared for Natural Resources Defense Council, San Francisco, California and Friant Water Users Authority, Lindsay, California by Stillwater Sciences, Berkeley, California.

Stillwater Sciences. 2006. Restoring recruitment processes for riparian cottonwoods and willows: a field-calibrated predictive model for the lower San Joaquin Basin. Prepared for CALFED Bay-Delta Ecosystem Restoration Program, Sacramento, California by J. Stella, Stillwater Sciences, Berkeley, California, in conjunction with J. Battles and J. McBride, Department of Environmental Science, Policy, and Management, University of California, Berkeley.

Strahan, J. 1984. Regeneration of riparian forests of the Central Valley. Pages 58–67 *in* R. E. Warner and K. M. Hendrix, editors. California riparian systems: ecology, conservation, and productive management. University of California Press, Berkeley.

Strange, E. M., K. D. Fausch, and A. P. Covich. 1999. Sustaining ecosystem services in human-dominated watersheds: biohydrology and ecosystem processes in the South Platte River basin. Environmental Management 24: 39–54.

Stromberg, J. C. 1997. Growth and survivorship of Fremont cottonwood, Goodding willow, and salt cedar seedlings after large floods in central Arizona. Great Basin Naturalist 57: 198–208.

Stromberg, J. C., D. T. Patten, and B. D. Richter. 1991. Flood flows and dynamics of Sonoran riparian forests. Rivers 2: 221–235.

Stromberg, J. C., B. D. Richter, D. T. Patten, and L. G. Wolden. 1993. Response of a Sonoran riparian forest to a 10-year return flood. Great Basin Naturalist 53: 118–130.

Stromberg, J. C., R. Tiller, and B. Richter. 1996. Effects of groundwater decline on riparian vegetation of semiarid regions: the San Pedro, Arizona. Ecological Applications 6: 113–131.

Stuart, J. D., and J. O. Sawyer. 2001. Trees and shrubs of California. University of California Press, Berkeley, California.

Suckling, K., D. N. Greenwald, and R. Silver. 1998. Petition to list the yellow-billed cuckoo *Coccyzus americanus* as a federally endangered species. Prepared by Southwest Center for Biological Diversity, Tucson, Arizona for U.S. Fish and Wildlife Service.

Sudworth, G. B. 1908. Forest trees of the Pacific slope. U.S. Government Printing Office, USDA Forest Service, Washington, D.C.

Szaro, R. 1989. Riparian scrubland and community types of Arizona and New Mexico. Desert Plants 9: 1–138.

Thompson, K. 1961. Riparian forests of the Sacramento Valley, California. Pages 294–315 *in* R. S. Platt, editor. Annals of the Association of American Geographers.

TNC (The Nature Conservancy). 2003a. Beehive Bend Subreach addendum to: A pilot investigation of cottonwood recruitment on the Sacramento River. The Nature Conservancy, North Central Valley Office, Chico, California

TNC (The Nature Conservancy). 2003b. Modeling plant community types as a function of physical site characteristics. The Nature Conservancy, Chico, California.

TNC (The Nature Conservancy). 2005. Restoring biodiversity health on the middle Sacramento River. The Nature Conservancy, Chico, California.

Trowbridge, W. B. 2002. The influence of restored flooding on floodplain plant distributions. Doctoral dissertation. University of California, Davis.

Tu, I.-Y. M. 2000. Vegetation and processes of natural regeneration in periodically flooded riparian forests in the Central Valley of California. Doctoral dissertation. University of California, Davis.

Vaghti, M. G. 2003. Riparian vegetation classification in relation to environmental gradients, Sacramento River, California. Master's Thesis, University of California, Davis.

Vaghti, M. G., and S. E. Greco. In press. Riparian vegetation of the Great Valley. In Terrestrial Vegetation of California, Third Edition.

Van Haverbeke, D. F. 1990. *Populus deltoides* var. *occidentalis* Rydb: Plains cottonwood. Pages 536–543 *in* R. M. Burns and B. H. Honkala, editors. Silvics of North America: hardwoods, Vol. 2. U.S. Department of Agriculture, Agriculture Handbook 654.

Walker, L R., and F. S. Chapin III. 1986. Physiological controls over seedling growth in primary succession on an Alaskan floodplain. Ecology 67: 1508–1523.

Whisenant, S. G. 1999. Repairing damaged wildlands: a process-oriented, landscape-scale approach. Biological conservation, restoration, and sustainability. Cambridge University Press, Cambridge, U.K.

Wiens, J. A. 2002. Riverine landscapes: taking landscape ecology into the water. Freshwater Biology 47: 501–515.

Wood, D. M. 2003a. Pattern of woody species establishment on point bars on the middle Sacramento River, California. Research and Long-term Monitoring Implementation: Beehive Bend and Chico Landing Sub-reaches. Prepared by Department of Biological Sciences, California State University, Chico for The Nature Conservancy, Sacramento River Project, Chico, California.

Wood, D. M. 2003b. The distribution and composition of woody species in riparian forests along the middle Sacramento River, California. Research and Long-term Monitoring Implementation: Beehive Bend and Chico Landing Sub-reaches. Prepared by Department of Biological Sciences, California State University, Chico for The Nature Conservancy, Sacramento River Project, Chico, California.

Figures

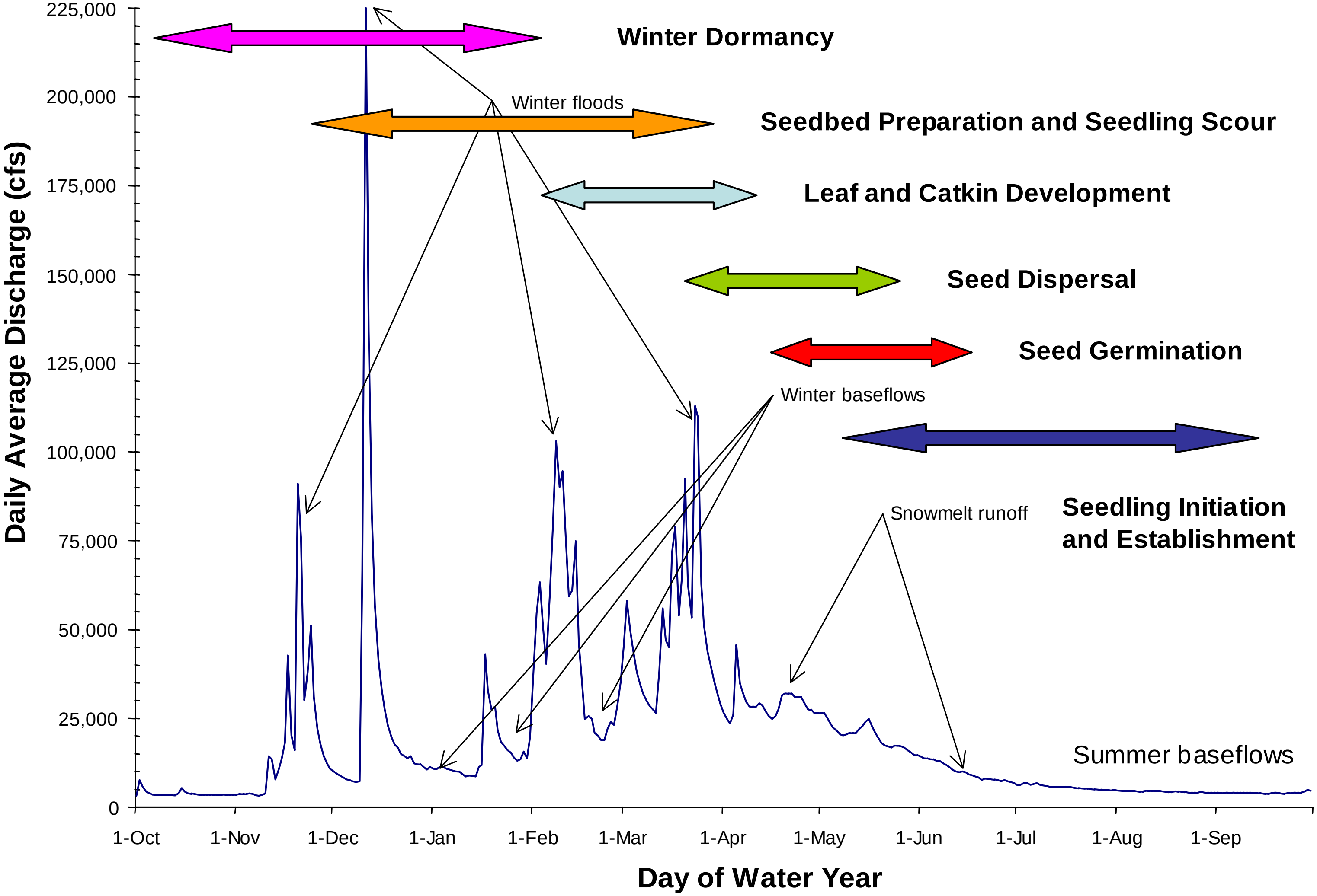


Figure 9-1. Representative annual hydrograph from the pre-Shasta period showing key hydrograph components relative to Fremont cottonwood life history timing. Hydrograph data are from the Bend Bridge Gate (near Red Bluff) for Water Year 1938 (Kondolf et al. 2000).

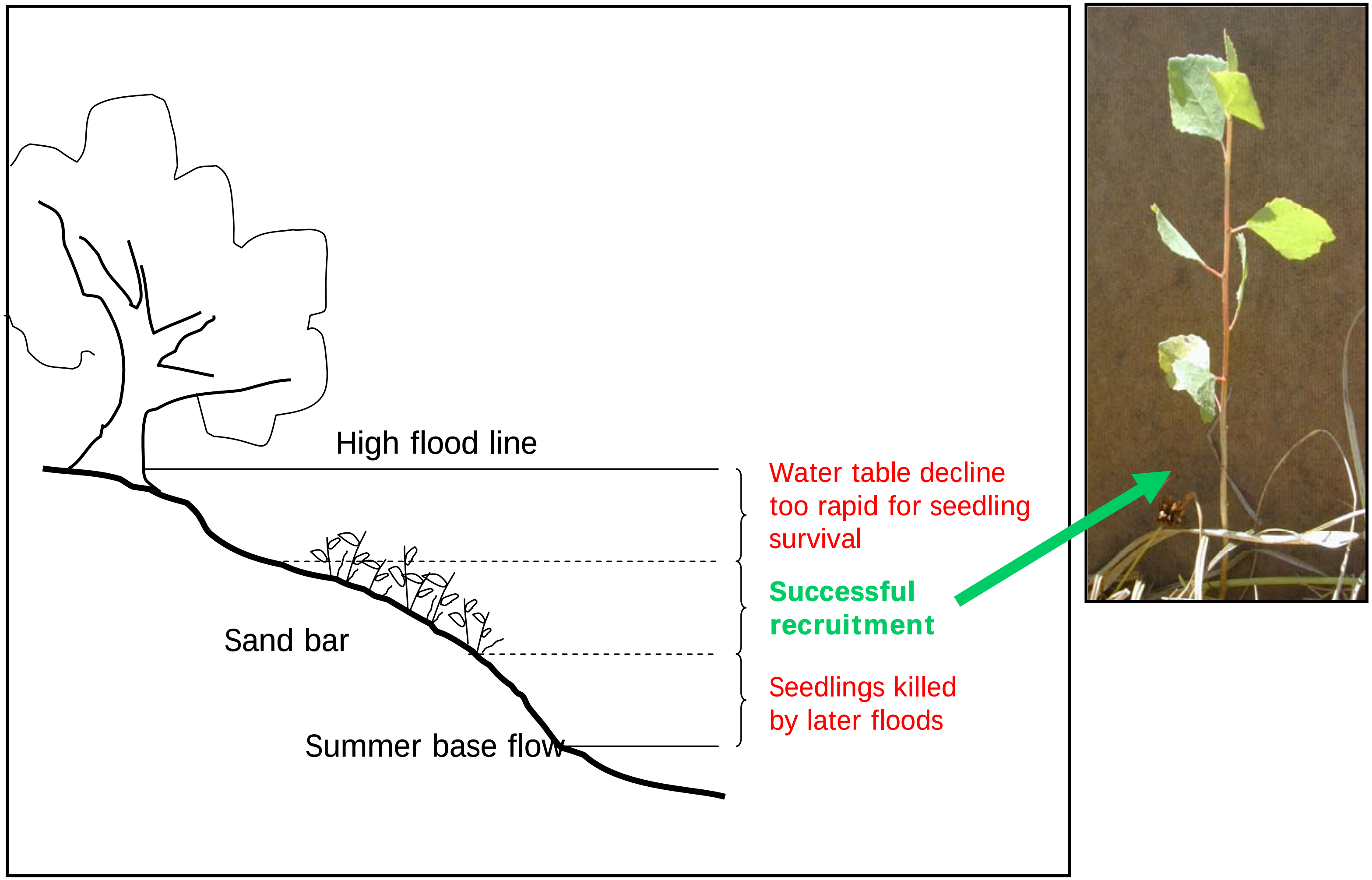


Figure 9-2. Generalized patterns of successful seedling recruitment observed for cottonwoods and willows along alluvial rivers. Seedlings that germinate too high on the bank cannot grow roots fast enough to keep up with the receding water table and soil moisture level during the hot summer months, while seedlings that initiate too low on the bank are removed by scour during high flow events during the subsequent winter or spring.

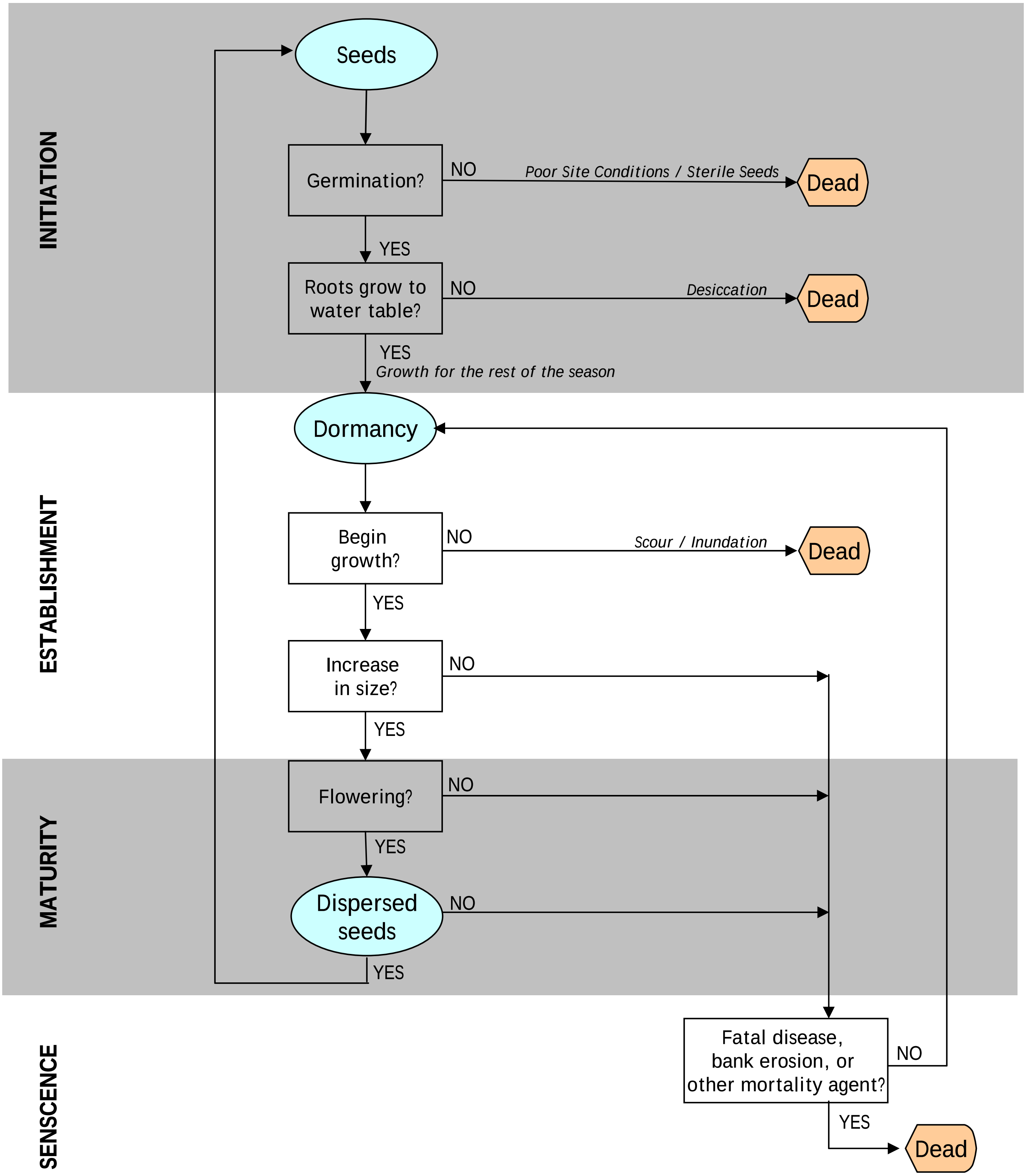
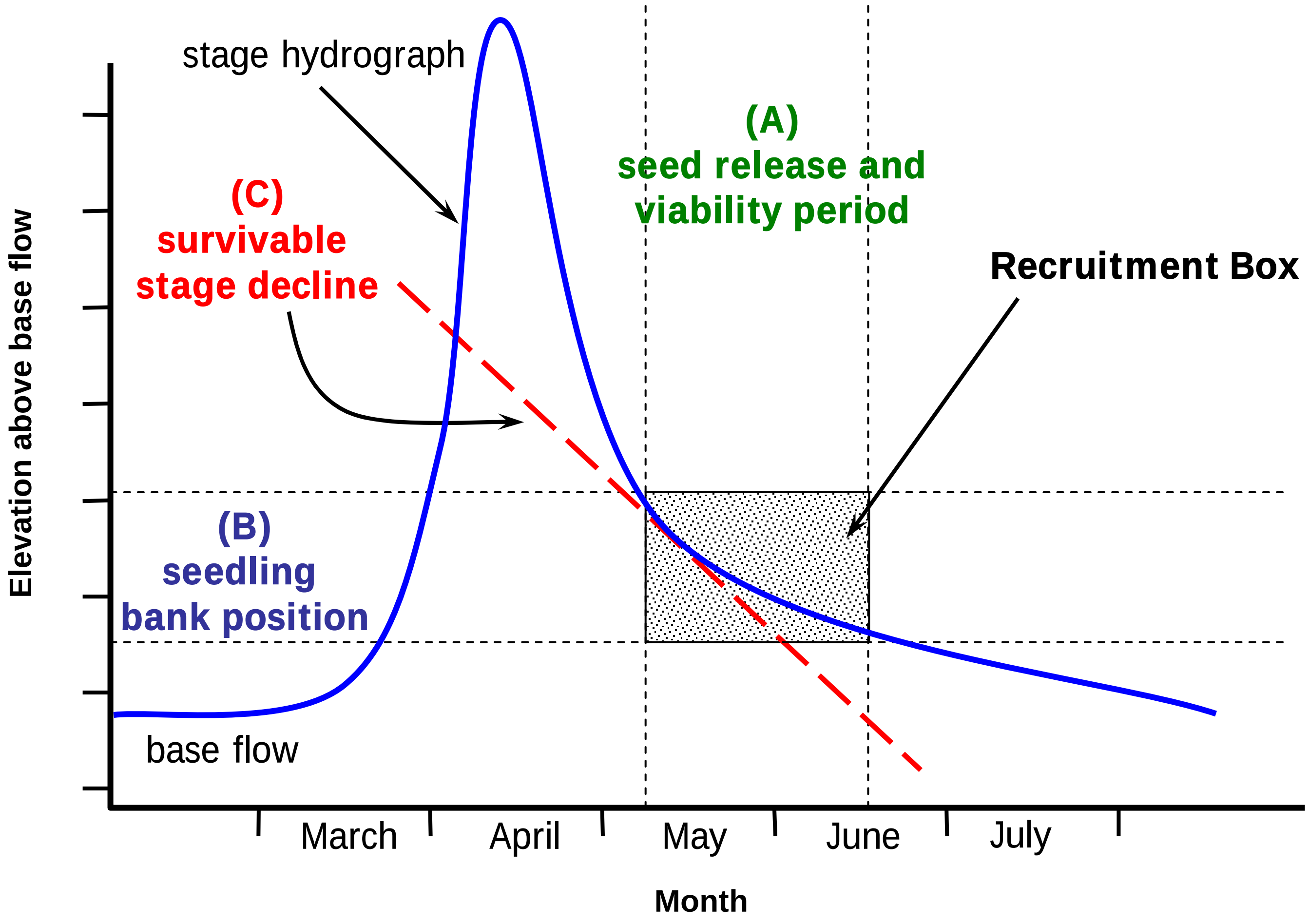


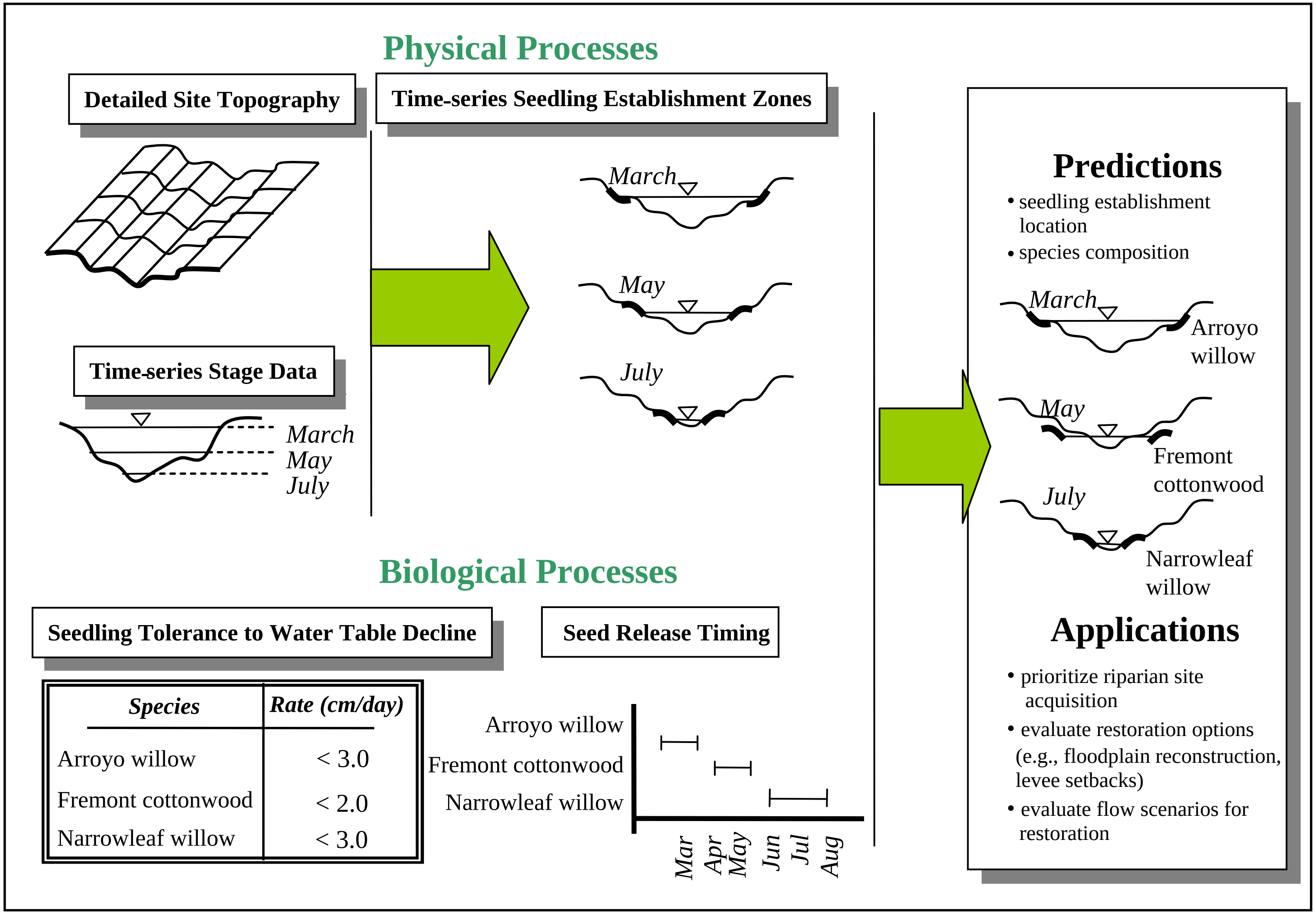
Figure 9-3. Generalized woody riparian plant life cycle, showing life stage and mortality agents that affect life stages. Modified from McBain & Trush (2002).



The 'Recruitment Box' Concept (redrawn from Mahoney and Rood, 1998)

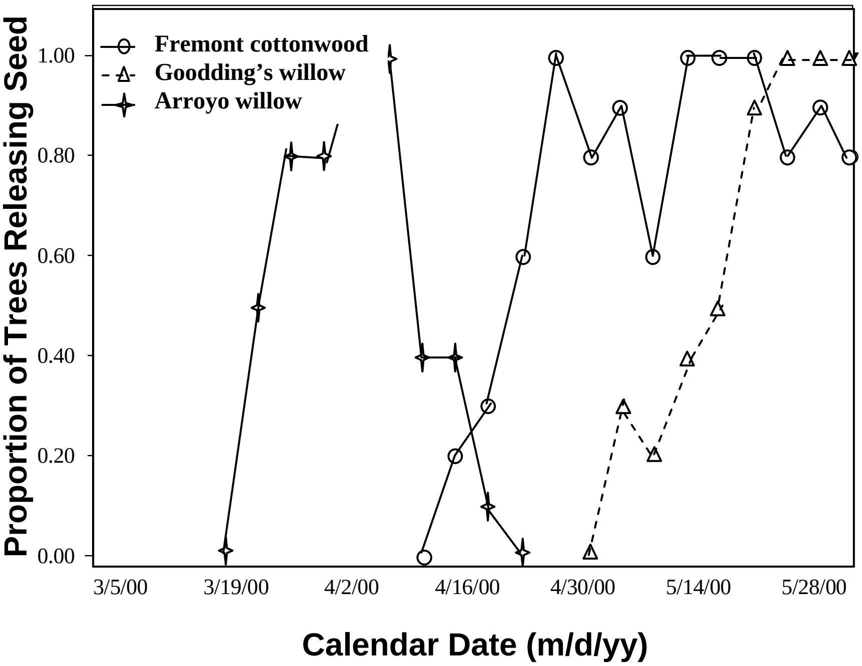
Figure 9-4. The recruitment box model, illustrating how (A) the window of seed release, dispersal and viability and (B) relative elevation above summer base flow (which defines the zone in which seedlings are not likely to dessicate in the summer or be scoured away during the winter) define the theoretical "recruitment box" conditions in which successful seedling initiation and establishment is possible if the stage recession rate of the spring hydrograph is equal to or slower than the survivable stage decline (C).

Application of Recruitment Box Model



Species	*Rate (cm/day)*
Arroyo willow	< 3.0
Fremont cottonwood	< 2.0
Narrowleaf willow	< 3.0

Figure 9-5. Framework for application of the recruitment box model to river corridor management and restoration.



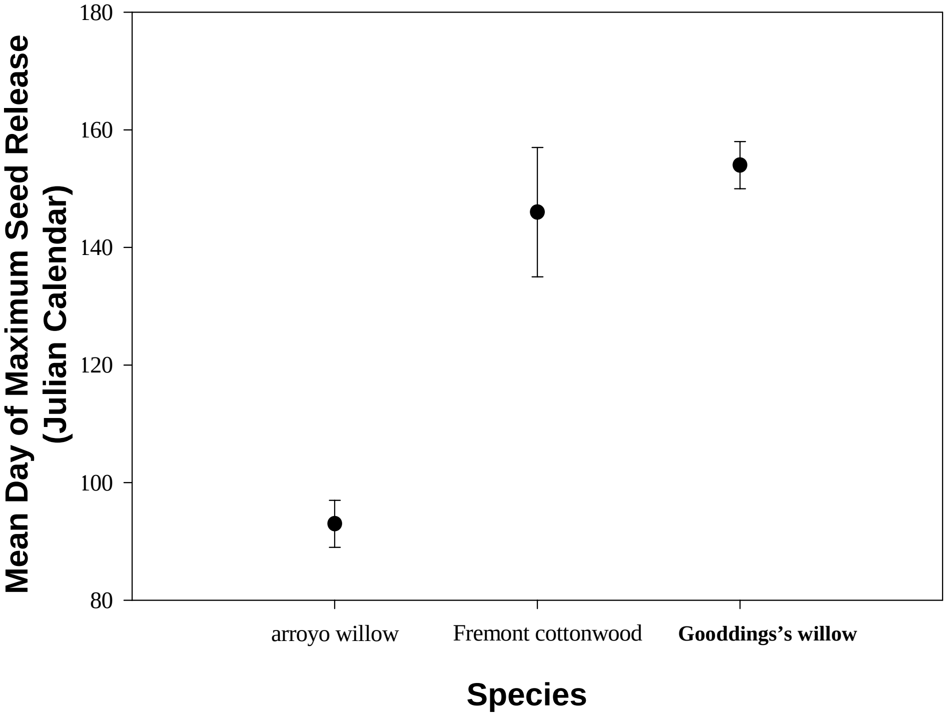


Figure 9-6. Seed release phenology for pioneer riparian tree species (top) and comparison of day of maximum seed release for pioneer riparian tree species (bottom) on the Sul Norte Unit of the SRNWR, spring 2000. Source: Peterson (2002).

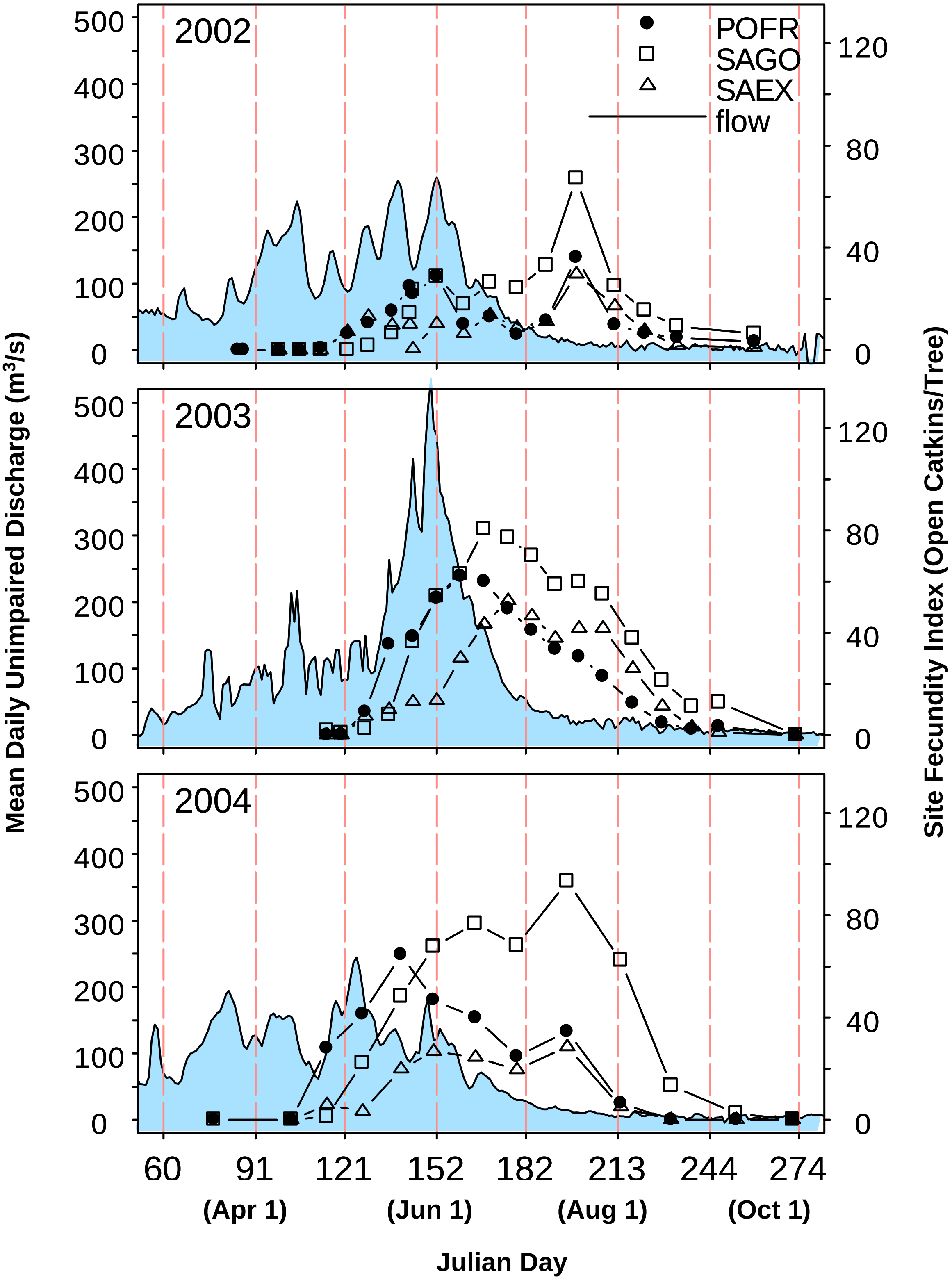


Figure 9-7. Patterns of seed release for Fremont cottonwood (Populus fremontii = POFR), Goodding's willows (Salix gooddingii = SAGO), and narrowleaf willow (S. exigua = SAEX) along the Tuolumne River compared to unimpaired flow for three years. For all species, the annual fecundity index is calculated as mean per-tree open catkin count for all surveys; data from all sites are pooled. The hydrograph is computed unimpaired discharge at La Grange Dam (California Data Exchange Center, gauge TLG). Source: Stella et al. (in press).

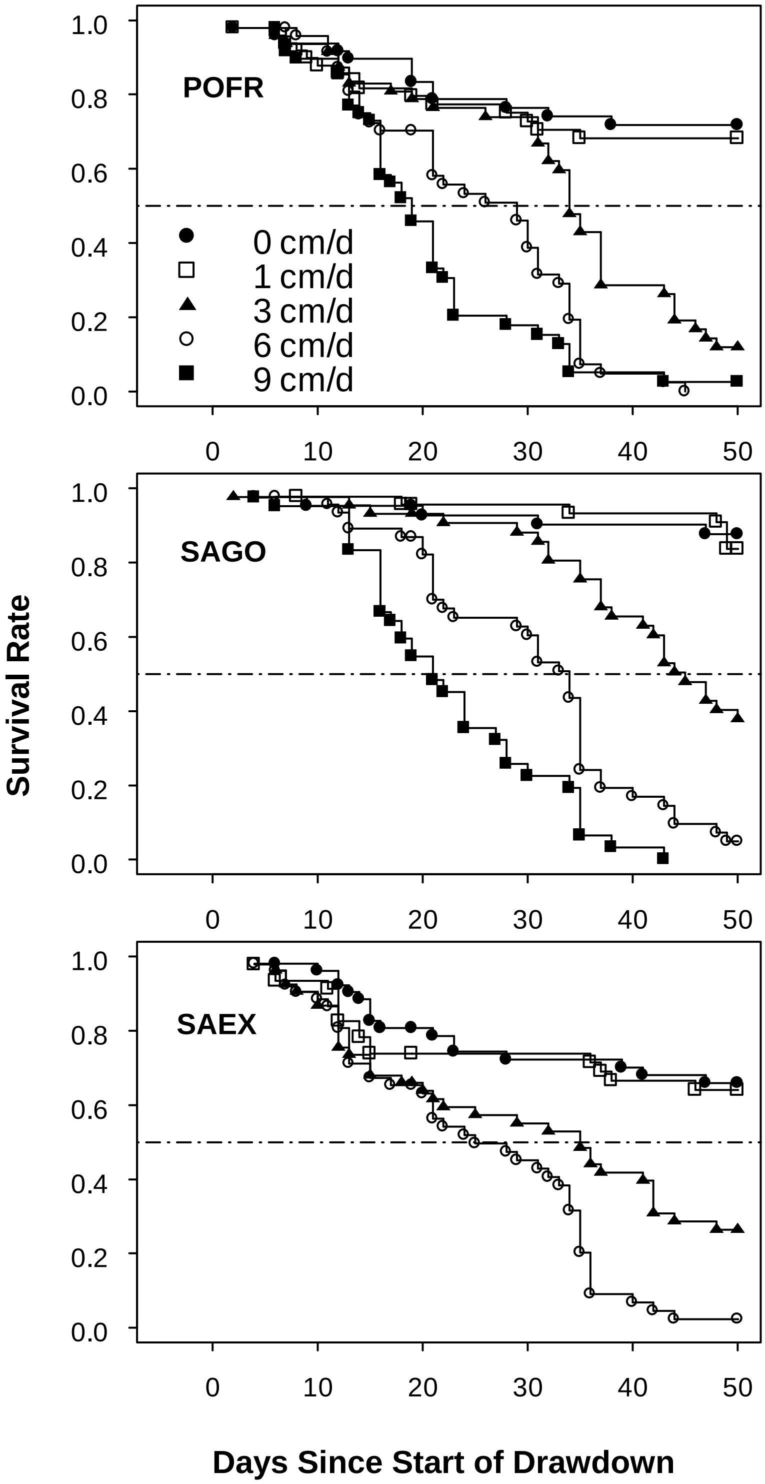


Figure 9-8. Seedling survival by species and treatment level (water level drawdown rate, ranging from 0 to 9 cm/day). Survival was estimated for censored data using the Kaplan-Meier method, which is a stepped function. Dashed line indicates median survival. POFR = *Populus fremontii* **(Fremont cottonwood), SAGO =** *Salix gooddingii* **(Goodding's willow), SAEX =** *Salix exigua* **(narrowleaf willow). Source: Stella et al. (in review) and Stillwater Sciences (2006).**

Figure 9-9. Seedling morphology at final harvest after a 60 day growth and survival experiment conducted by Stella et al. (in review) and Stillwater Sciences (2006). Fremont cottonwood seedlings are at upper left, Goodding's willow at right, and narrowleaf willow at lower left. Each seedling is a single representative individual from any treatment group with at least one survivor at the end of the experiment. Treatment groups are ordered from left to right for each species in the following sequence: control (0 cm/day) and 1, 3, and 6 cm per day. No cottonwood or narrowleaf willow seedlings survived from the 6 cm d-1 treatment group, and no seedlings of any species survived the 9 cm/day treatment. The black and white bars at left and right indicate 1 cm increments.

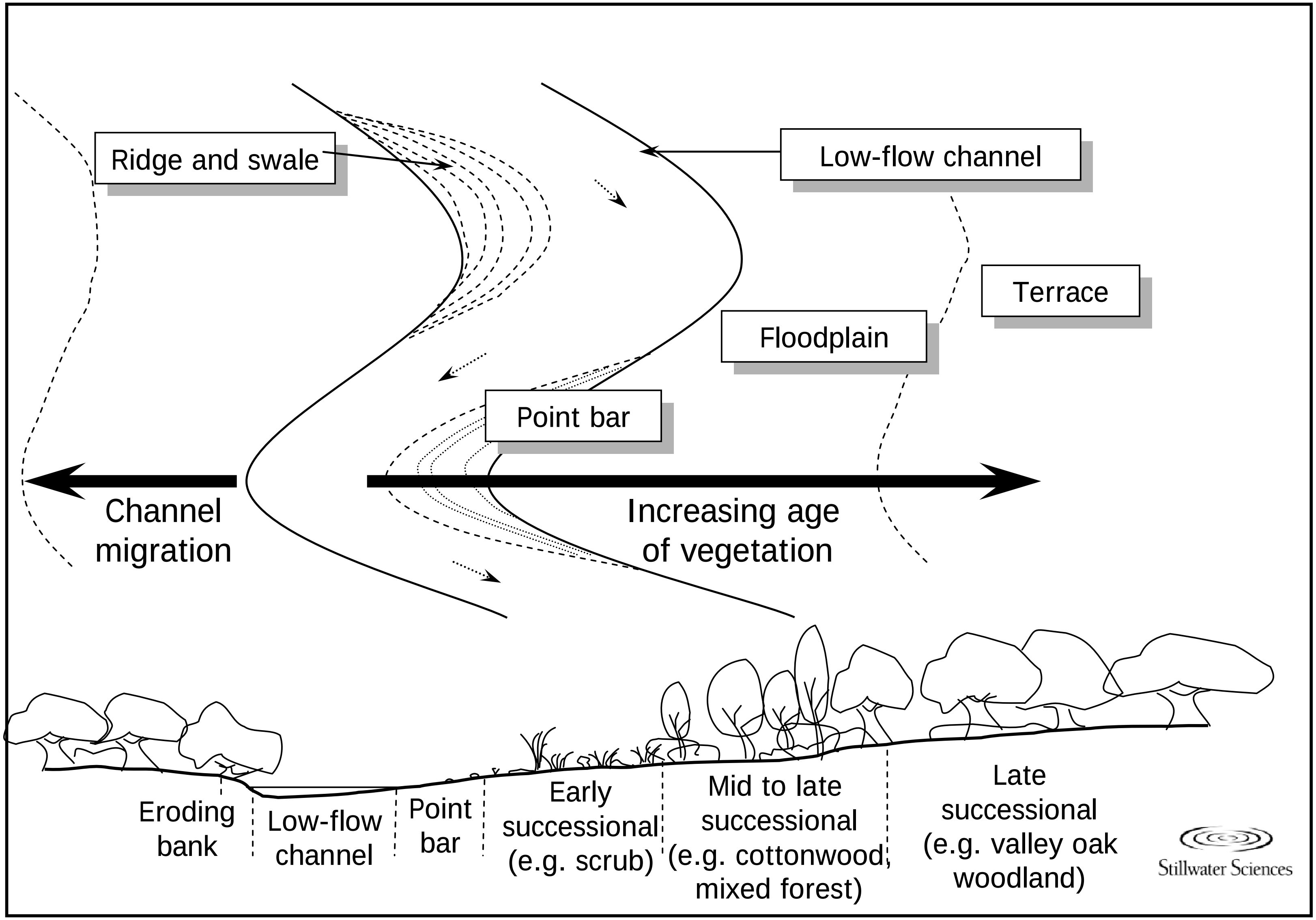


Figure 9-10. Schematic diagram of riparian succession conceptual model for meandering alluvial rivers.

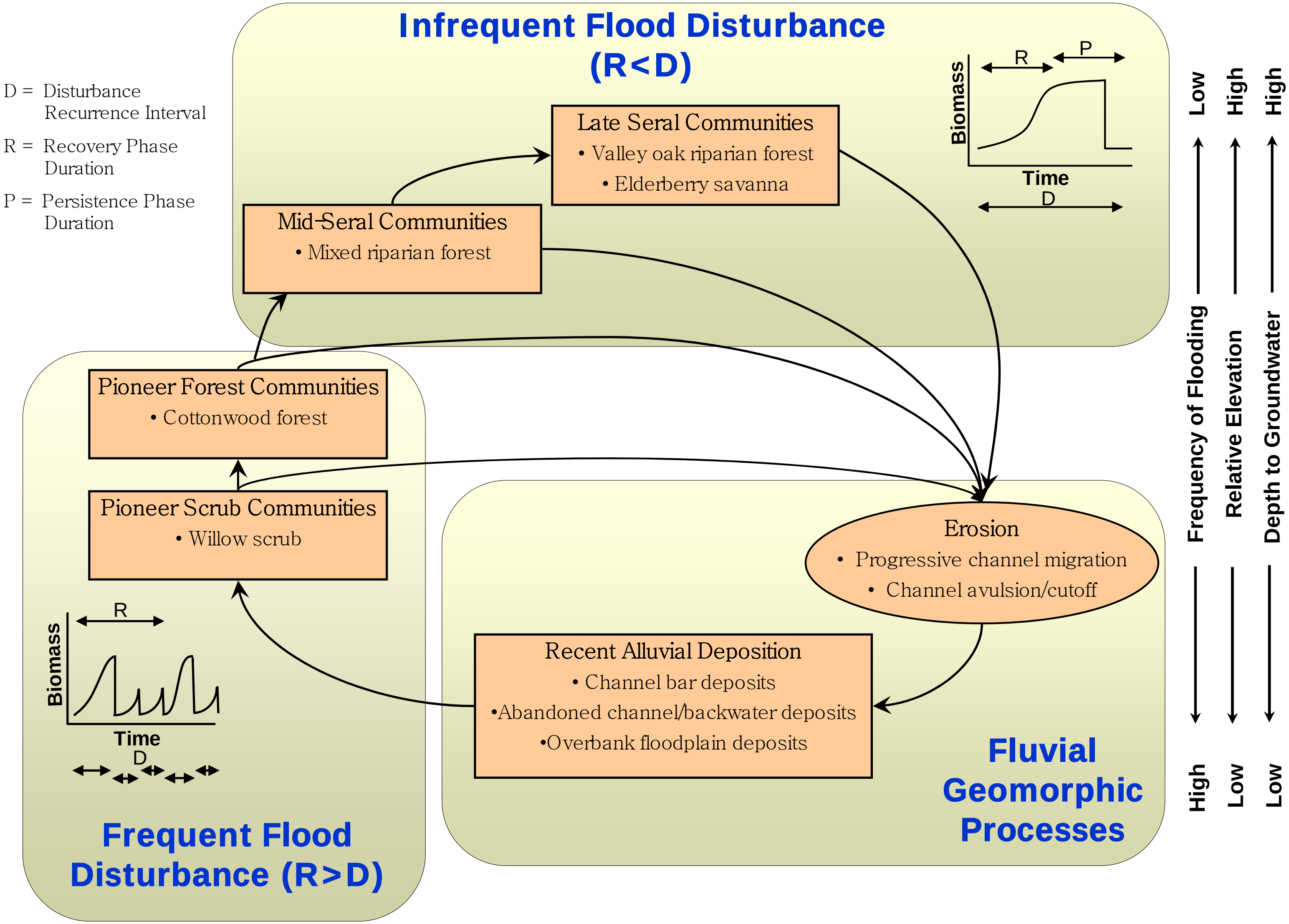


Figure 9-11. Conceptual model of the influence of flood disturbance and fluvial geomorphic processes (sediment transport, erosion, deposition) on riparian vegetation succession.

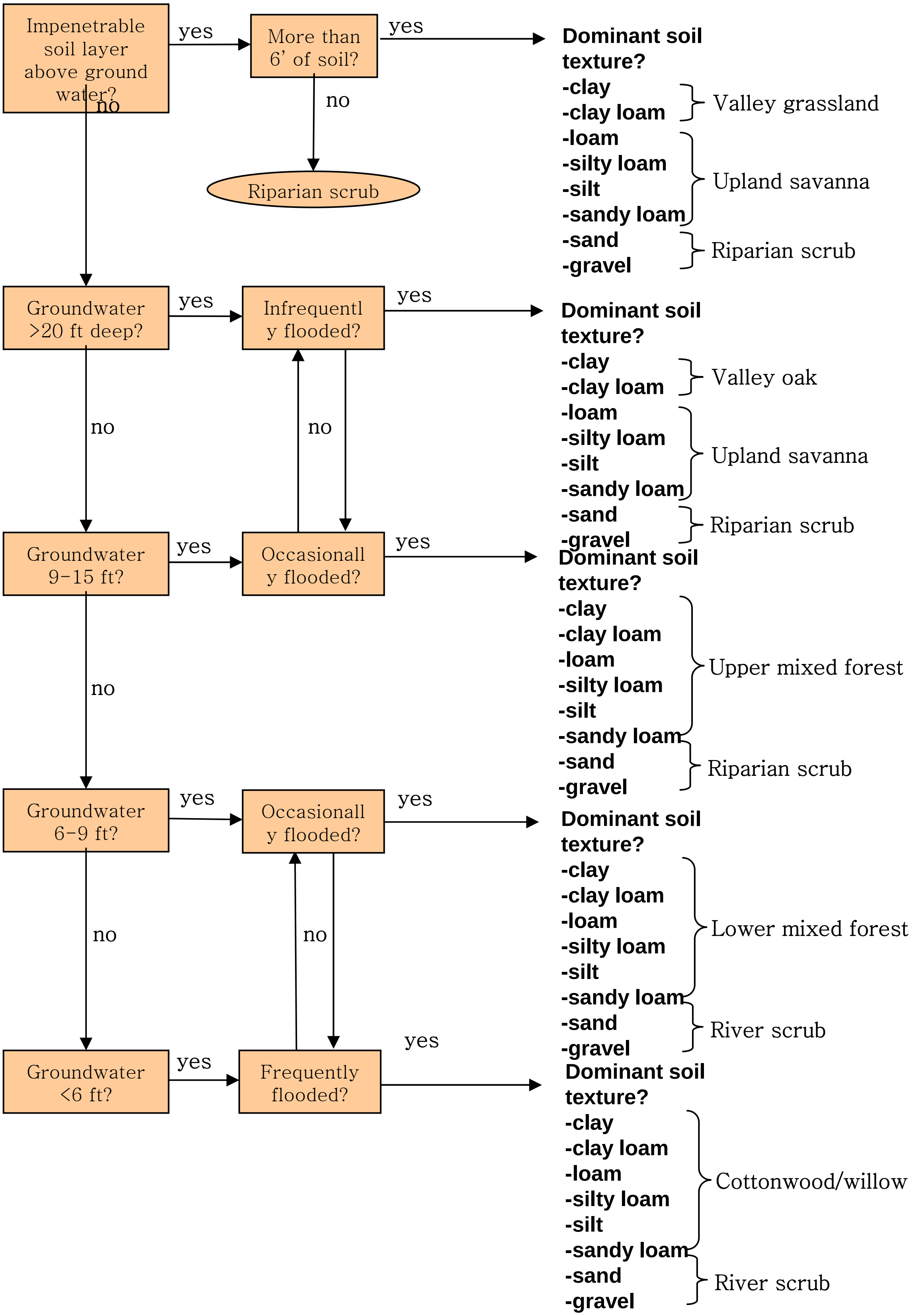


Figure 9-12. Preliminary plant design model showing effects of soil depth and flood frequency on vegetation community type. Source: TNC 2003b.

COTTONWOOD COMMUNITY ECOSYSTEM MODEL

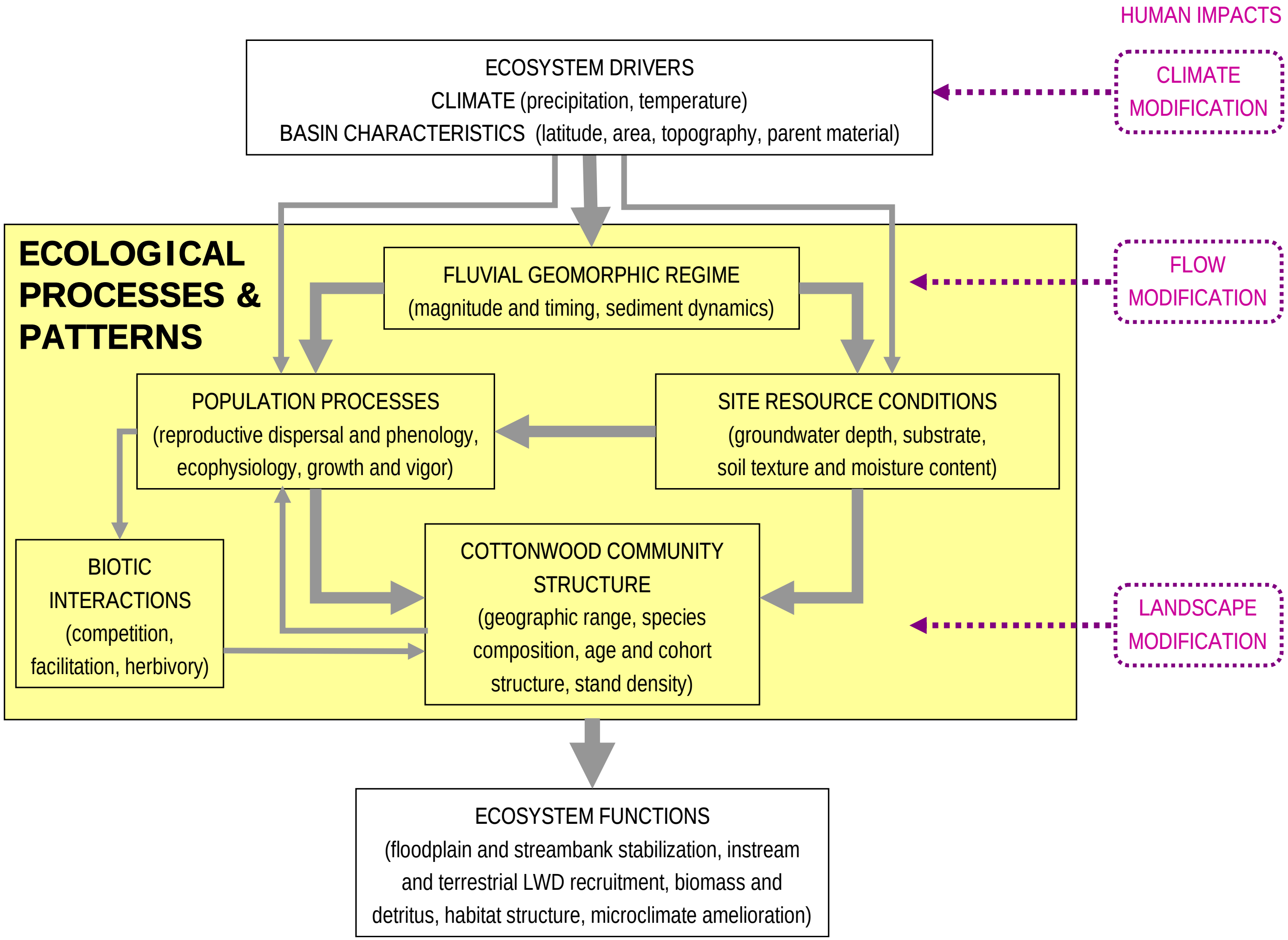


Figure 9-13. Conceptual model of natural and anthropogenic factors affecting the dynamics of cottonwood forest community development, structure, and ecosystem functioning in the middle Sacramento River corridor. Adapted from Battles et al. (2005), based on the framework developed by Strange et al. (1999).

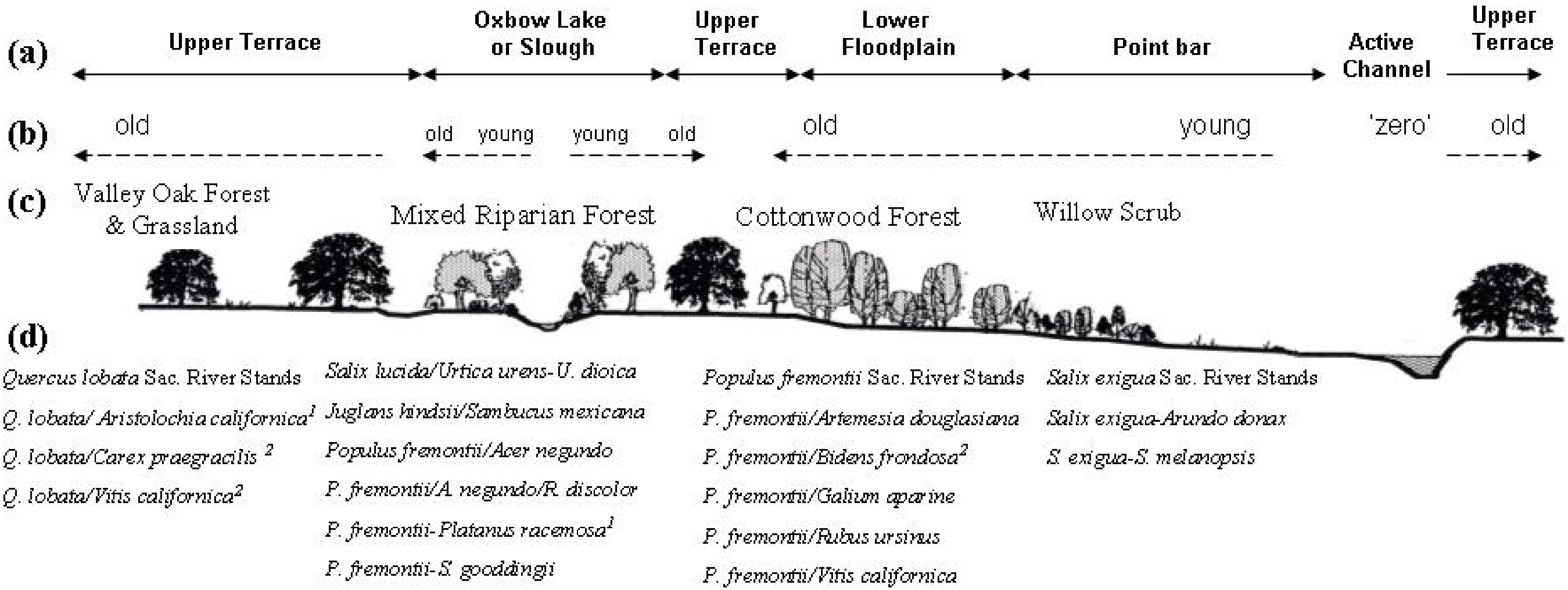


Figure 9-14. Conceptual model showing general pattern of native riparian vegetation relative to the river channel and floodplain features. a = Geomorphic feature, b = relative age of vegetation, c = primary vegetation type, d = plant alliances and associations. Source: Greco et al. (in review).

Impacts on Riparian Vegetation

Floodplain development

Habitat fragmentation

Channelization and bank revetment

Figure 9-15. Examples of anthropogenic alterations to Central Valley river-riparian systems.

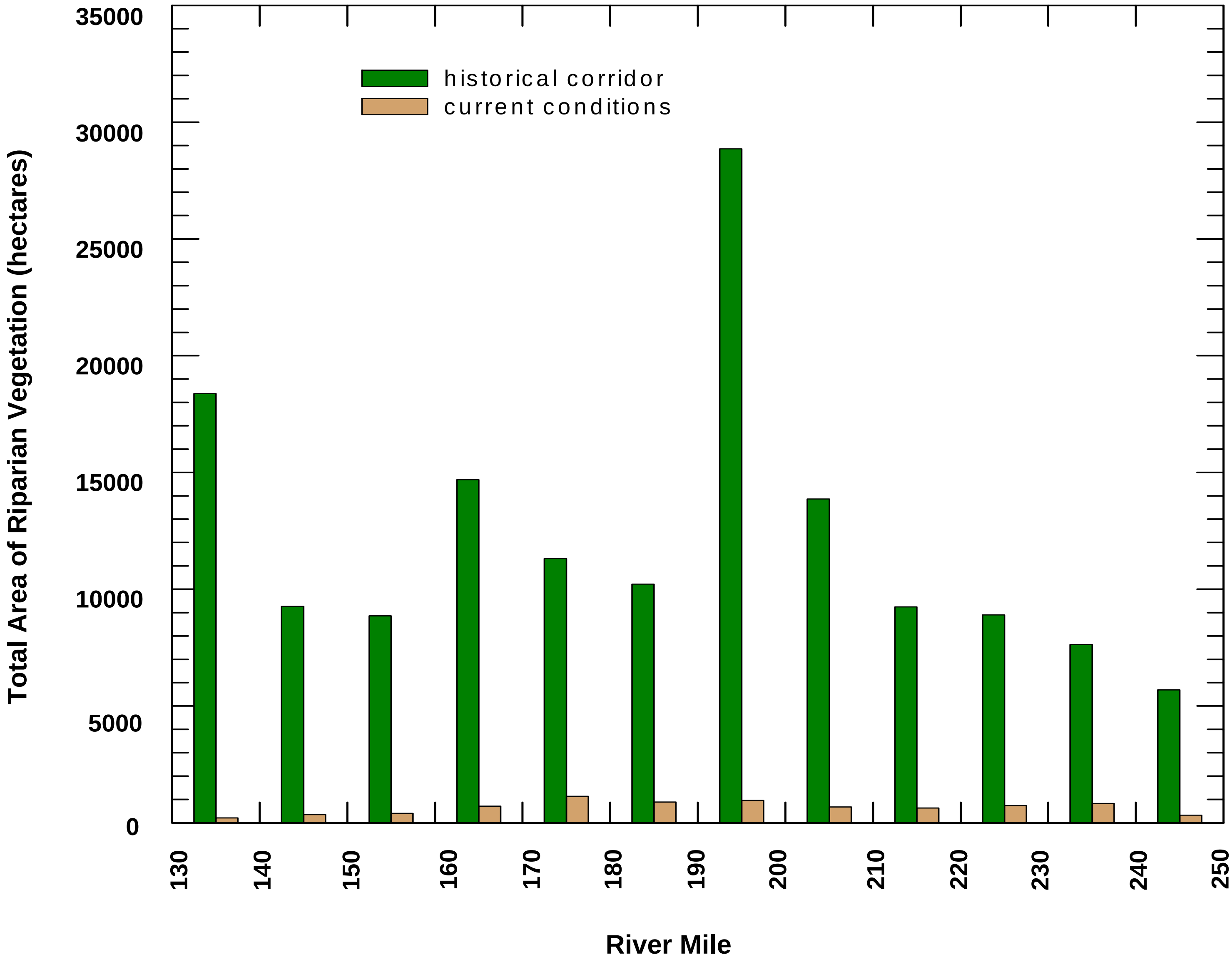


Figure 9-16. Comparison of the extent of the estimated historical riparian zone (reconstructed from soil survey data) with the current amount of riparian vegetation along the Sacramento River corridor from River Mile (RM) 130 through RM 250, by 10-mile intervals. Source: Stillwater Sciences analysis using Chico State University GIS vegetation data.

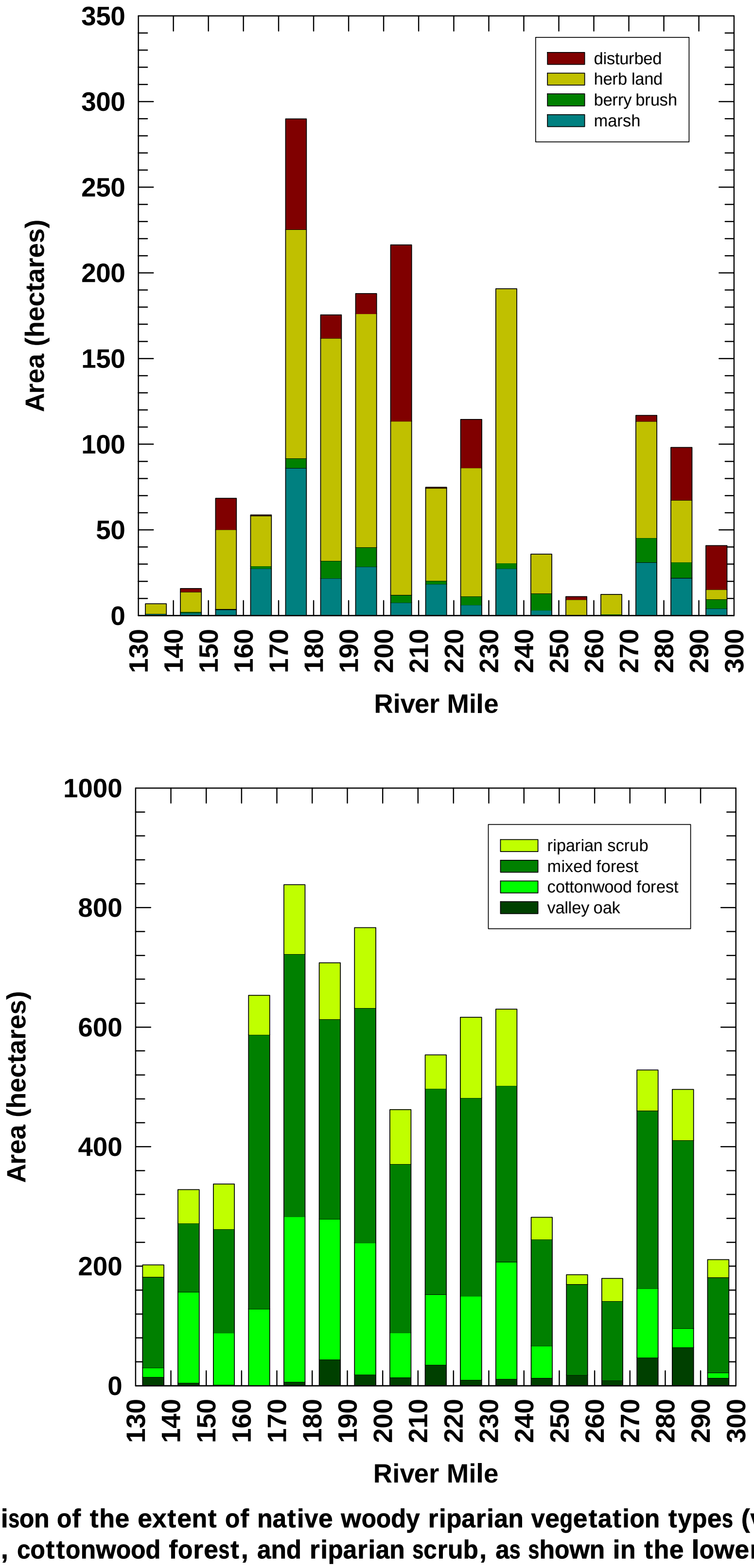
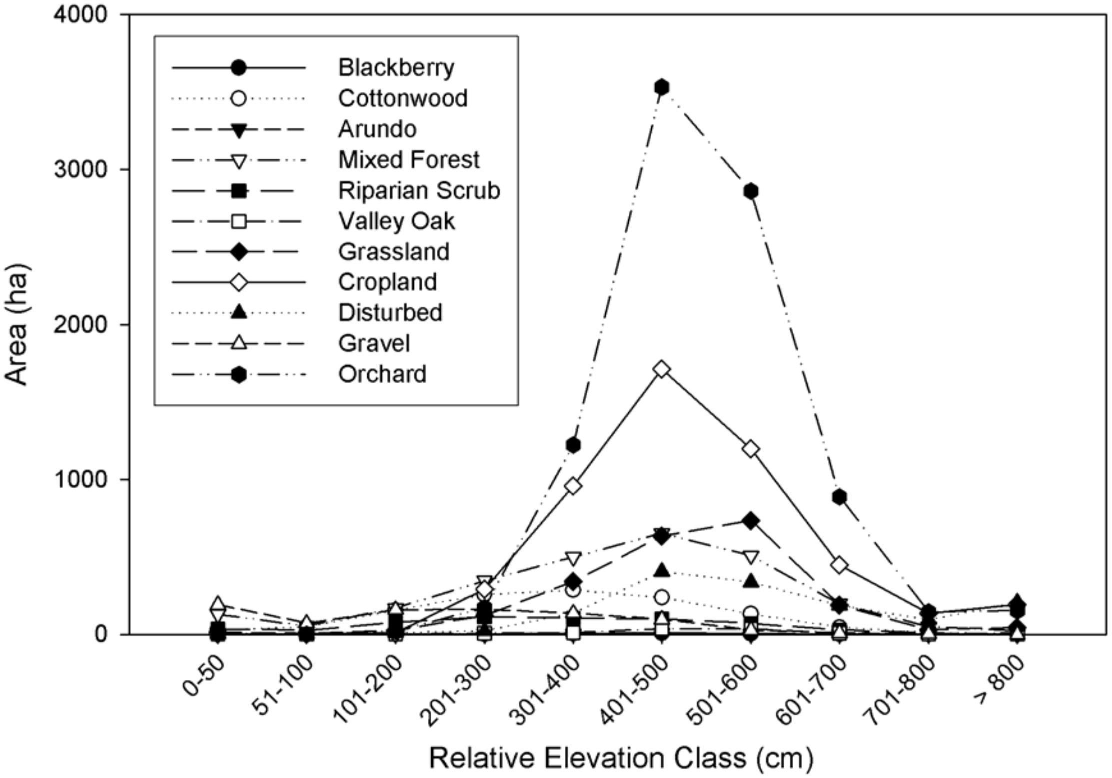


Figure 9-17. Comparison of the extent of native woody riparian vegetation types (valley oak woodland, mixed riparian forest, cottonwood forest, and riparian scrub, as shown in the lower plot) (top) with other types of riparian vegetation (marsh, herb land, berry scrub, disturbed, as shown in the upper plot) (bottom) along the Sacramento River corridor from River Mile (RM) 130 through RM 250, by 10-mile intervals. Source: Stillwater Sciences analysis using Chico State University GIS vegetation data.



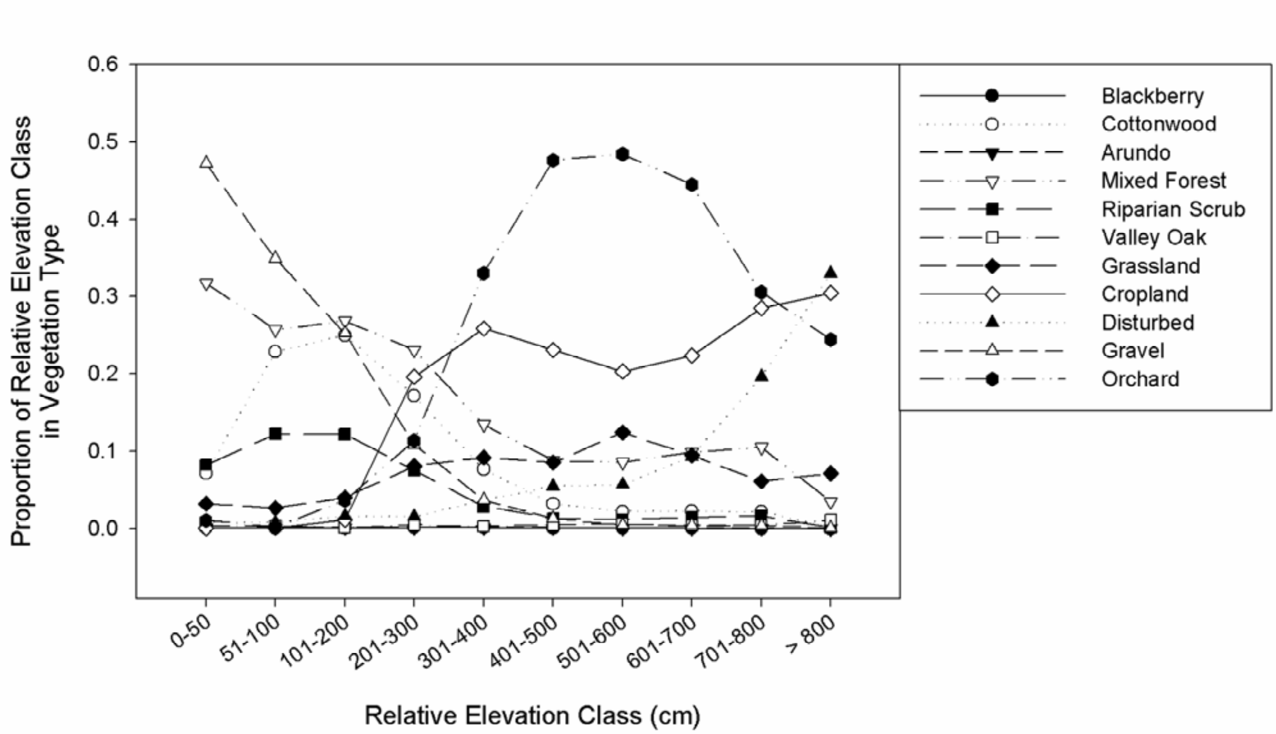


Figure 9-18 a, b. The proportion of the total area of each relative elevation class covered by each vegetation type. Note that in Figure B, the proportions add up to 1.0 for each relative elevation class. Source: Greco et al. (in review).

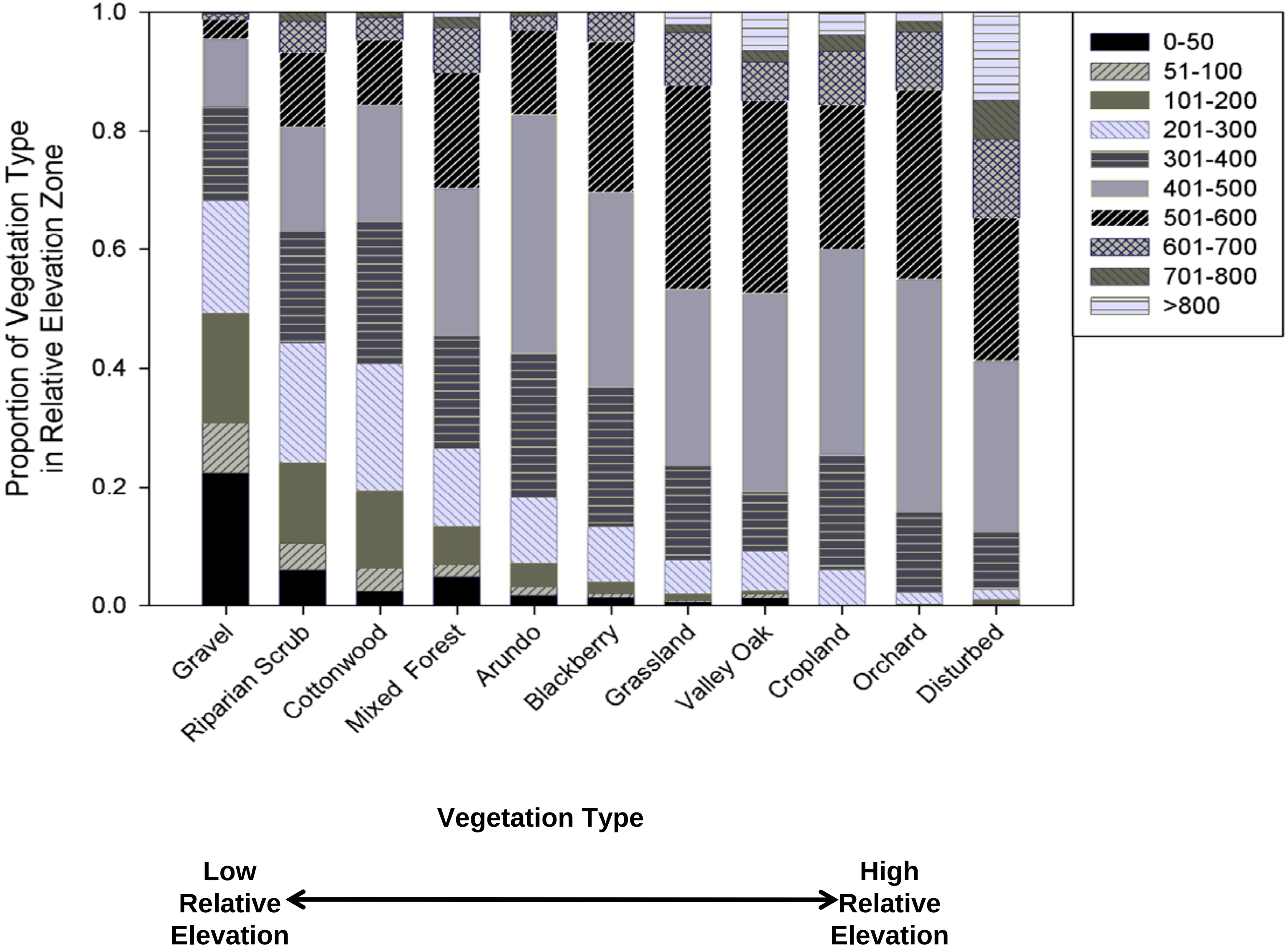


Figure 9-19. The proportion of the total area of each vegetation type that is in each relative elevation size class (ranging from 0 to 800 cm [0 to 315 in]) above mean summer baseflow elevation. Note that the vegetation types tending to occur at lower relative elevations are located toward the left side of the graph, and vegetation or land covers found more often in higher relative elevation classes are located toward the right. Source: Greco et al. (in review).

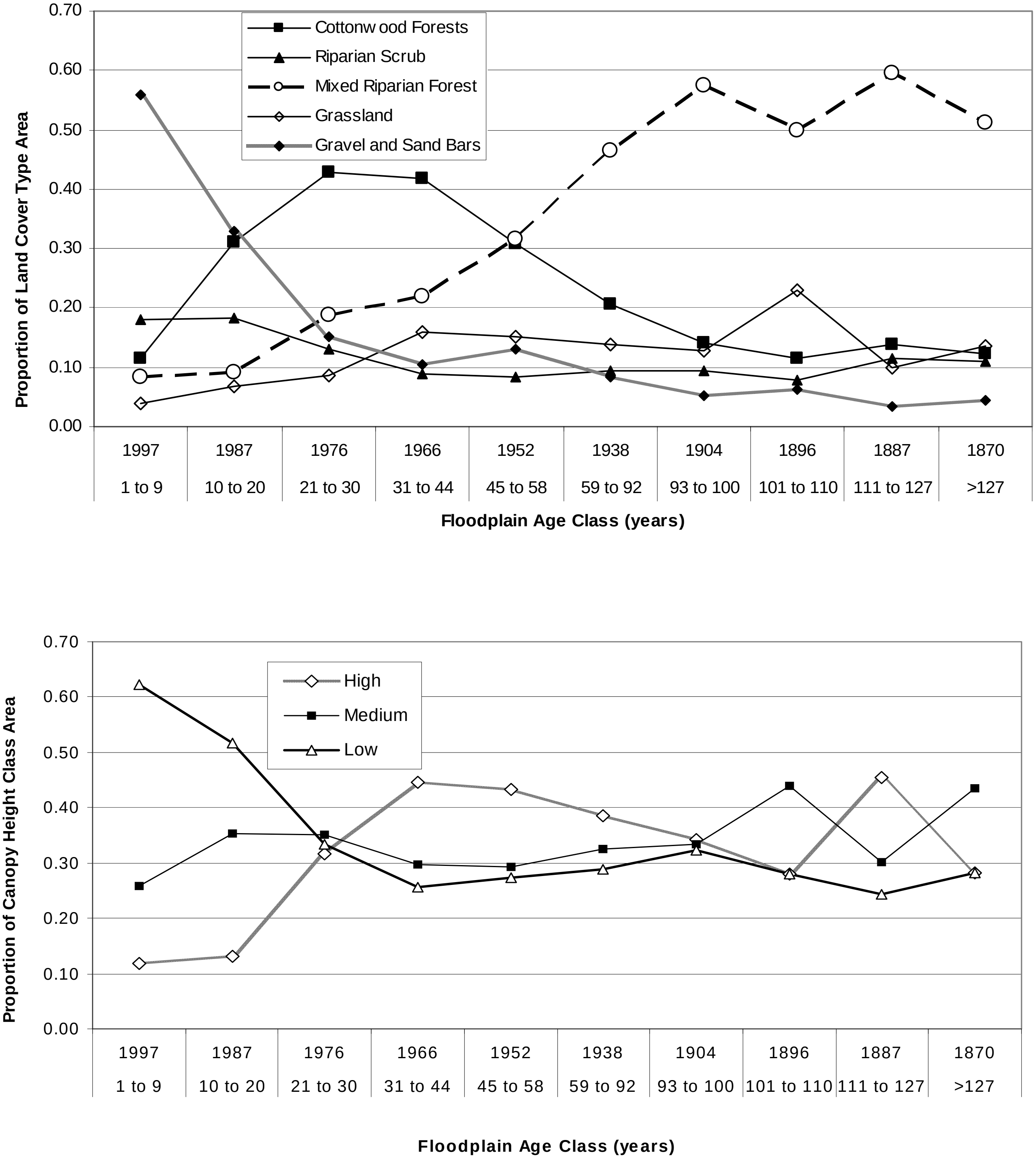
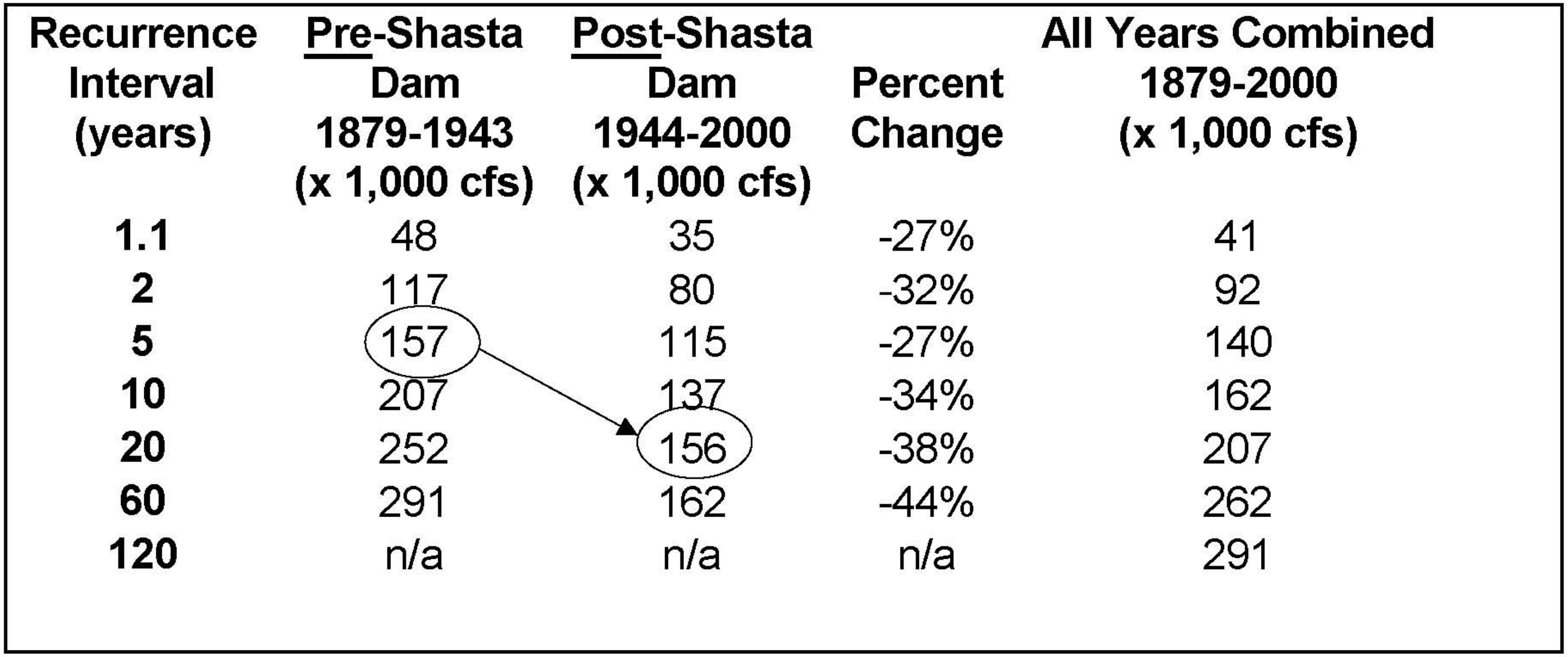
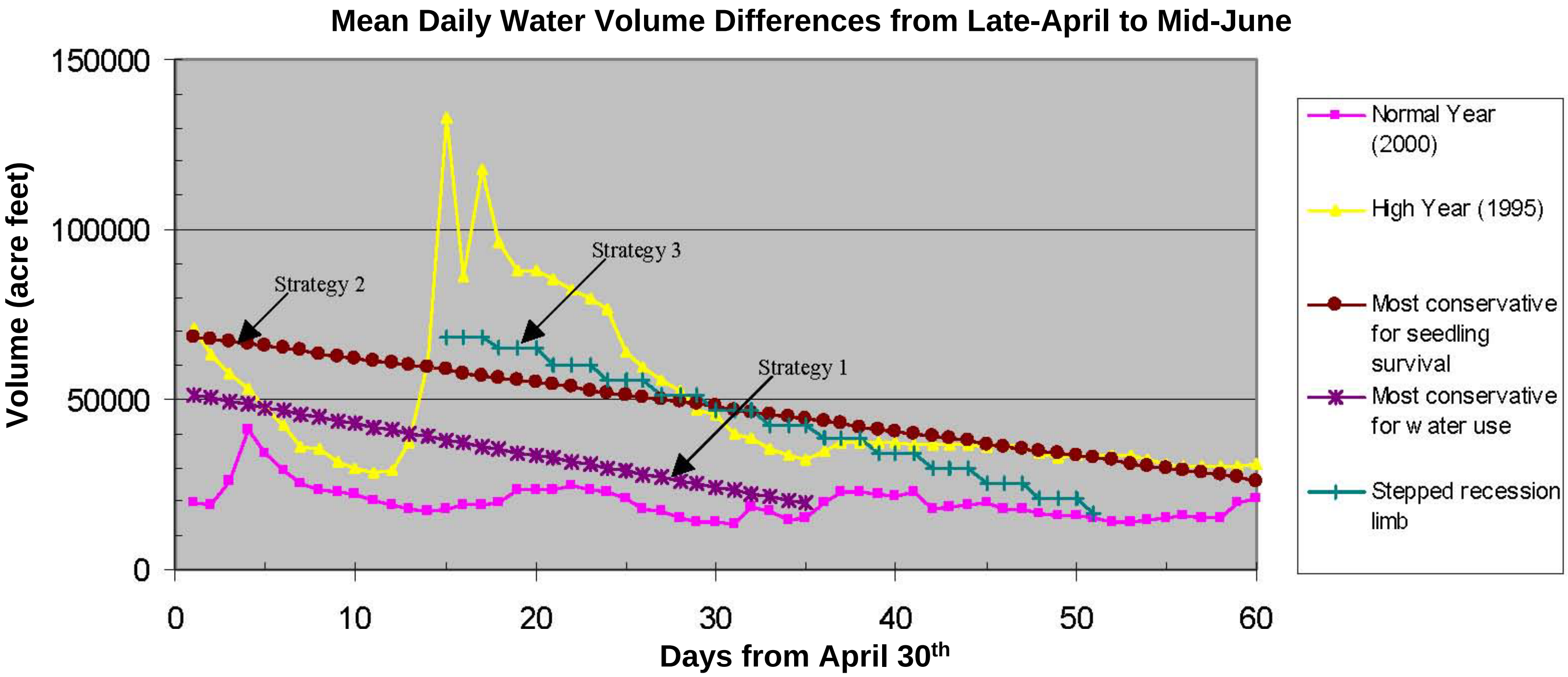


Figure 9-20 a, b. Proportion of land cover types and canopy height classes by floodplain age class. Source: Greco et al. (in review).



Recurrence Interval (years)	Pre-Shasta Dam 1879-1943 (x 1,000 cfs)	Post-Shasta Dam 1944-2000 (x 1,000 cfs)	Percent Change	All Years Combined 1879-2000 (x 1,000 cfs)
1.1	48	35	-27%	41
2	117	80	-32%	92
5	157	115	-27%	140
10	207	137	-34%	162
20	252	156	-38%	207
60	291	162	-44%	262
120	n/a	n/a	n/a	291

Figure 9-21. Recurrence interval analysis at Bend Bridge, 1879-2000. This analysis suggests that a pre-Shasta 5-year recurrence interval flood of 155,000 to 160,000 cfs now occurs less frequently, at a recurrence interval of approximately 20 years. Less frequent recruitment of cottonwood riparian forests may result from this hydrologic alteration. Source: Lowney and Greco 2003 and TNC 2003.



Options for Cottonwood Recruitment Flows

Strategy	Drawdown rate (ft/day)	Beginning elevation	Reference year	Total volume of application (ac ft)	Already available	Managed difference (volume needed - already available) (ac ft)	% change from existing flow
1	0.108	117.0	2000	1,239,505	667,485	572,020	46%
1	**0.108**	**117.0**	**1995**	**1,239,505**	**1,944,979**	**-705,474**	**-37%**
2	0.083	119.0	2000	2,845,693	1,131,661	1,714,032	60%
2	**0.083**	**119.0**	**1995**	**2,845,693**	**1,911,880**	**933,813**	**4%**
3	stepped	119.0	2000	1,632,710	698,588	934,122	57%
3	**stepped**	**119.0**	**1995**	**1,632,710**	**2,011,667**	**-378,957**	**-23%**

Figure 9-22. Recruitment flow strategies and total volume of water required. The strategies are designed to promote cottonwood recruitment at three cottonwood point bar study sites (RM 192, 183, and 172). Strategy 3, which uses a stepped recession limb, represents a compromise between required water volume and seedling survival that would use 23% less water than the amount that flowed by the study sites in 1995. Source: TNC 2003.

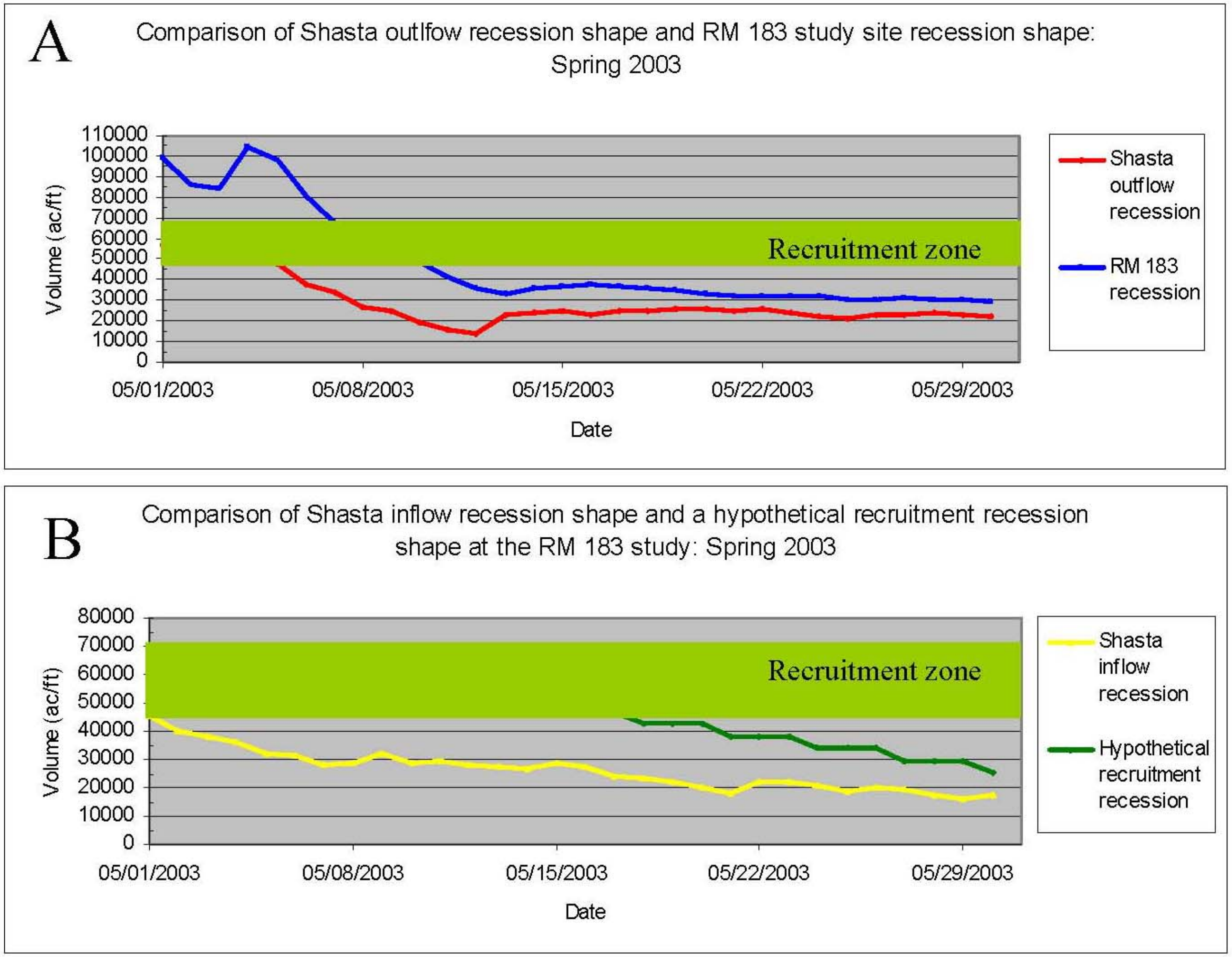


Figure 9-23. Comparison of Spring 2003 hydrograph recession limb shapes for the RM 183 cottonwood study site and Shasta Reservoir inflow and outflow. The top graph (A) shows similarities in the pattern of outflow from Shasta Reservoir and the recorded flow at RM 183. The lower graph (B) shows the similarity between natural inflow (yellow) pattern to Shasta Reservoir and a hypothetical recruitment flow (green) designed to promote cottonwood recruitment at the RM 183 point bar study site. Source: TNC 2003.

10 SYNTHESIS AND RECOMMENDATIONS

This report uses six focal species (Chinook salmon, steelhead, green sturgeon, bank swallow, western pond turtle, and Fremont cottonwood) to organize a discussion of ecological processes and habitats that occur in the mainstem Sacramento River. The preceding chapters address the habitat needs of the different life history stages of each focal species. Collectively, the habitat needs of these six species cover the major habitat types (e.g., off-channel water bodies, gravel riffles, point bars, floodplain) that occur in the Sacramento River corridor.

This chapter synthesizes information from the previous chapters to draw general conclusions about management activities, ecological processes, and habitats for each focal species and, in particular, highlights issues that are likely to influence the wellbeing of multiple species. These general conclusions provide the foundation for recommendations designed to restore ecological processes and habitats that will benefit not only the focal species profiled in this report, but also other species with similar habitat requirements. These recommendations include research and monitoring activities that can address some of the fundamental uncertainties and data limitations that complicate management and restoration of the Sacramento River corridor. Some of the suggested studies are also designed to test hypotheses that spring from the conceptual models contained in the focal species chapters of this report.

10.1 Synthesis/General Conclusions

10.1.1 Continued loss of salmonid spawning and rearing habitat

Chapters 3 through 5 discuss how human activities eliminated a significant amount of historical habitat for salmonids in the Sacramento River basin. Large water supply dams caused the largest loss of habitat by preventing upstream passage. For example, the primary spawning populations of winter-run Chinook salmon and late-fall-run Chinook salmon in the Sacramento River basin lost access to all historical habitat when Shasta Dam was completed (Clark 1929, Yoshiyama et al. 2001). Similarly, spring-run Chinook salmon lost access to over 70% of historical spawning habitat in the Central Valley (Yoshiyama et al. 2001). Though fall-run Chinook salmon did not lose as great a *percentage* of historical habitat, the population likely lost the greatest *amount* of spawning habitat when Shasta Dam was completed.

Initially, the loss of historical spawning habitat was ameliorated for several salmonids by the change in the water temperature regime caused by Shasta Dam operations. By storing and releasing colder water in the alluvial reach of the mainstem channel, the dam created more favorable water temperature conditions where there were more abundant gravel resources, as compared with upstream and tributary reaches (Slater 1963). All salmonid populations that spawned historically above Shasta Dam, except fall-run Chinook salmon, likely enjoyed a cumulative, though temporary, *increase* in spawning habitat as they were displaced downstream of the dam. For example, historical winter-run spawning habitat was likely limited in the tributaries above Shasta Dam, confined to high-elevation reaches fed by volcanic springs that maintained water temperatures below 56°F (13°C) throughout the summer so that eggs could incubate successfully. These high elevation stream reaches are generally more narrow, with steeper gradients, than the broad alluvial reaches located downstream. As a result, these upstream reaches generally have less gravel to support spawning because in-channel gravel storage is limited to short slope breaks between high gradient reaches or local pockets of gravel associated with structures that reduce flow velocities and induce sediment deposition (e.g., boulders, LWD). In contrast, the alluvial reaches of the mainstem Sacramento River offered winter-run salmon large gravel riffles, so the change in the water temperature regime increased the extent of suitable spawning habitat available to the population. This initial increase in spawning habitat contributed to higher escapements of winter-run Chinook salmon until the early 1970s (estimated escapements exceeded 100,000 adults in 1969), after which run sizes rapidly declined (Yoshiyama et al. 1998, p. 506, Table 5). However, as detailed in Chapters 3 and 4, bed coarsening below the dam eventually reduced the spawning habitat available to the winter-run population, thereby contributing to declines in escapements.

State and federal agencies have augmented the gravel supply of the upper Sacramento River to compensate for the lost gravel supply that is now trapped behind Shasta Dam (Chapters 3 and 4). Since 1978, nearly 250,000 yd^3 (191,100 m^3) of spawning-sized gravel has been added to the mainstem Sacramento River between Keswick Dam (RM 302) and the confluence with Clear Creek (RM 290). This gravel has played an important role in maintaining local patches of spawning habitat located near the injection sites. However, the scale of gravel augmentation to date is small when compared to the cumulative deficit of more than 10 million yd^3 (7.6 million m^3) of coarse sediment that were mined from the channel and floodplains or trapped by Shasta Dam since the early 1940s. Though the added gravel has helped to *maintain* remnant patches of spawning habitat, it has likely done little to *expand* spawning habitat significantly and reverse the process of bed coarsening.

Spawning habitat availability in the mainstem Sacramento River affects all salmonid populations, but winter-run and fall-run Chinook salmon are particularly vulnerable to spawning habitat limitations because of their life history strategies (Chapter 4). Both runs depend on the production of large numbers of fry that migrate downstream in pulses soon after emergence, thereby swamping predators to combat predation mortality. The production of a large fry component requires ample spawning gravel. Limitations in spawning habitat can increase competition for suitable gravels among spawning adults, inducing redd superimposition that increases egg mortality and, in turn, reduces fry production. Similarly, poor quality spawning gravels (e.g., framework gravels with a high percentage of fine sediment) can reduce survival-to-emergence, which can also reduce fry production. Reductions in fry production can undermine the swamping effect of small fry migrating downstream in schools, thereby increasing predation risk. As a result, the population effects of spawning habitat limitations can be non-linear, because the population loses potential recruits not only from the eggs that fail to survive, but also from a greater percentage of fry lost to predation.

In contrast, steelhead, spring-run salmon, and late-fall-run salmon populations have a more substantial fraction of juveniles that oversummer in the river before emigrating as larger juveniles or smolts. By emigrating downstream at a larger size, these juveniles can better avoid predators because of improved swimming ability and because they use deeper channel habitats than fry, which typically use channel margins. Larger juveniles may also grow larger than the gape of some predators, further reducing predation risk. Because we would expect relatively higher rates of juvenile survival for steelhead, spring-run Chinook, and late-fall-run Chinook smolts (Fisher 1994), these populations can likely be sustained with fewer spawning gravel resources than those required by fall- and winter-run Chinook populations. Rearing habitat, rather than spawning habitat, is more likely to be the primary density-dependent limiting factor for these salmonids in the Sacramento River.

By changing the water temperature regime in the alluvial reach of the Sacramento River, Shasta Dam likely increased the area of spawning habitat available to spring-run Chinook salmon as compared with historical conditions in the upstream tributaries. However, this increase was temporary not only because of progressive bed coarsening that occurred in the post-dam era, but also because Shasta Dam eliminated the spatial segregation of spring-run and fall-run salmon spawning. Historically, spring-run salmon were able to access higher elevation tributaries because adults migrated upstream during periods of snowmelt, which provided passage over flow-related obstacles. In contrast, fall-run salmon were usually limited to lower-elevation reaches because they migrated upstream during periods of low flow. Because there is little temporal segregation between spring-run and fall-run spawning, the spatial segregation was essential for keeping the runs distinct. When both runs were displaced downstream of Shasta Dam, the spatial segregation of spawning was eliminated and hybridization between the two runs was inevitable. Hybridization was also caused by hatchery practices that inadvertently mixed the two stocks (NMFS 2004). The loss of spatial segregation generally conferred an advantage on fall-run Chinook in the Sacramento River because they spawn later than spring-run Chinook. Eggs deposited by spring-run adults in August and September were susceptible to mortality from redd superimposition when fall-run spawning activity peaked in October and November. In essence, the loss of spatial segregation with fall-run Chinook reduced the amount of *effective* spawning habitat available to spring-run Chinook, in addition to the loss of physical habitat caused by progressive bed coarsening, in the post-dam era.

The life history strategy of spring-run Chinook salmon does not make them as susceptible to spawning habitat limitations as the winter-run and fall-run populations, assuming they have access to appropriate spawning grounds. Nevertheless, the combined effects of spawning habitat loss and hybridization with fall-run Chinook have virtually eliminated the spring-run population as a distinct run in the mainstem Sacramento River (Lindley et al. 2004). Without some mechanism to re-constitute the spatial segregation between fall-run and spring-run salmon spawning, the spring-run population is likely to receive little benefit from any restoration or management measures to increase spawning habitat availability in the upper Sacramento River.

Agency reports released in the mid-1990s suggested that spawning habitat was not likely a limiting factor for the depressed populations of steelhead and Chinook salmon in the Sacramento River at the time (USFWS 1995, NMFS 1997). This assessment seems to be based, in part, on surveys of salmonid spawning gravels that estimated over 1.1 million ft^2 (0.1 million m^2) of suitable gravel area between Keswick Dam (RM 302) and Jelly's Ferry (RM 267) in the late 1980s and early 1990s (USFWS 1995). However, the area of mapped spawning *gravel* does not necessarily indicate the amount of spawning *habitat* that is available, because other environmental variables (e.g., water depth, water velocity, interstitial flow, downwelling) influence habitat quality and use. Also, the 1987 survey assessed surface gravels, which does not account for gravel depth, which can also influence spawning habitat availability and quality. Similarly, microhabitat conditions (e.g., water depth, water velocity, downwelling) can change with discharge. The 1987 gravel survey provides a valuable assessment of spawning habitat potential between Keswick Dam and Jelly's Ferry (Vogel and Taylor 1987), but it does not convert readily into a measure of spawning habitat area. Additional consideration of other environmental variables that influence spawning habitat quality (e.g., gravel depth, water depth, flow velocity) would suggest that there was less spawning habitat available than the 1.1 million ft^2 (0.1 million m^2) of spawning gravel that was mapped between Keswick Dam and Jelly's Ferry in 1987.

Using the results of the 1987 gravel survey, we can estimate the number of salmon that were supported by available habitat by making some simplifying assumptions. Estimates of Chinook salmon redd size vary considerably (Healey 1991), but Burner recorded Chinook salmon redd sizes in the Columbia river basin ranging from 42 to 69 ft^2 (3.9 m^2 to 6.5 m^2) (Burner 1951). Research conducted in the Tuolumne River suggests that fall-run Chinook salmon construct redds that average approximately 50 ft^2 (4.6 m^2) (EA Engineering 1992), which we can use as a coarse estimate of redd size. Because a female salmon will defend a constructed redd from other potential spawners for 4–25 days before succumbing (Healey 1991), the area required for spawning is greater than the redd size. Researchers in the Central Valley have often used a multiplier of four when converting redd size to defended area, citing Burner (1951), which suggests that 200 ft^2 (18.6 m^2) are required for each redd. Research conducted in the Tuolumne River suggests that salmon redds can be distributed more densely, with defense area of approximately 100 ft^2 (9.3 m^2) (EA Engineering 1992). We can use both of these assumptions to calculate a range of salmon escapements that could have been supported by the gravel mapped in 1987 (Vogel and Taylor 1987). If we assume conservatively that 1 million ft^2 (92,903 m^2) of spawning habitat was available between Keswick Dam (RM 302) and Jelly's Ferry (RM 268) in 1987 (of the 1.1 million ft^2 (102,193 m^2) of spawning gravel mapped by Vogel and Taylor), then roughly 5,000 redds could be accommodated if a defense area of 200 ft^2 (18.6 m^2) is assumed, and roughly 10,000 redds could be supported if we assume a defense area of 100 ft^2 (9.3 m^2).

Estimates of maximum spawning potential are tempered by the fact that redds are not perfectly distributed in suitable habitat. Adult salmonids often bypass seemingly suitable habitat to spawn in more upstream habitats. For example, winter-run Chinook salmon spawn primarily above Airport Road Bridge (RM 284) (CDFG 2004), and redds are often packed densely in this reach even though there is suitable spawning habitat and water temperatures located father downstream. Fall-run Chinook salmon exhibit the same pattern in the Tuolumne River, where they bypass downstream gravel riffles that appear to provide suitable spawning habitat to spawn densely in riffles located closer to La Grange Dam (EA Engineering 1992). There are several possible factors that may contribute to a preference for upstream spawning habitats. Many salmonid populations in the Central Valley have been displaced from historical habitats by large water supply dams that force spawning to occur downstream of the dam. Salmonids may be genetically hard-wired to migrate as far upstream as possible to spawn, so that spawning occurs most densely in riffles near the limit of anadromy. Microhabitat features that affect spawning habitat quality (e.g., interstitial flow, downwelling) may also render bypassed habitats less suitable than they appear from the surface (Vronskiy 1972). The upstream spawning preference may also reflect selective pressure applied by periods of elevated water temperature that sporadically eliminate the progeny of downstream spawners.

The upstream spawning preference exhibited by salmonids in the Sacramento River has important implications for the management and restoration of salmonid spawning habitat. The first implication is that efforts to restore salmonid spawning habitat will likely yield greater benefits for multiple salmonids if applied in upstream reaches. Application of gravel injection in upstream reaches will also address bed coarsening where it is most acute. Agencies have already recognized this implication by focusing past gravel augmentation efforts in the reach between Keswick Dam (RM 302) and Clear Creek (RM 290) (Bigelow 1996, Buer 1985).

A second implication is that efforts to significantly expand spawning habitat through gravel augmentation may not create habitat in proportion to the effort and resources expended. Salmon select spawning locations based on several factors, including gravel depth, interstitial flow, downwelling, and local hydraulics that can vary spatially depending on discharge. It is difficult to re-create microhabitat conditions that affect spawning habitat utilization (e.g., downwelling) through the direct placement of gravel in the channel. Similarly, a percentage of the gravel that is introduced to the channel, whether placed directly in the channel or staged on channel banks for recruitment by high flows, will deposit in areas that will not support spawning. It is important to emphasize, however, that we believe gravel augmentation is a necessary and valuable restoration approach for maintaining, increasing, and improving salmonid spawning habitat; indeed, in Chapter 10.2.2 we recommend that gravel augmentation be implemented on a more significant scale than has been practiced to date. Coarse sediment is a fundamental building block of aquatic habitats, and the disruption of the sediment supply since the completion of Shasta Dam has reduced and degraded salmonid spawning habitat in the Sacramento River. The point here is to caution against simplifying assumptions that a volume of gravel will yield an equivalent area of spawning habitat, because such equations can lead to activities that emphasize satisfying the minimal requirements of a species. The real focus of a gravel augmentation program is to increase in-channel storage of coarse sediment, so that the river has sufficient material to build and maintain habitats.

A third implication of the upstream spawning preference exhibited by salmonids in the Sacramento River relates to water temperature management, which is discussed below.

10.1.2 Changing the water temperature regime

Currently, one of the principal dedications of environmental water in the Sacramento River is designed to maintain suitable water temperatures to support the spawning and egg incubation of endangered winter-run Chinook salmon. The USBR operates the Trinity and Shasta divisions of the Central Valley Project to comply with SWRCB Order No. 90-5 and a 1993 Biological Opinion (BO) rendered by NMFS (NMFS 1993). SWRCB Order No. 90-5 requires USBR to maintain average daily water temperatures below 56^{o}F (13°C) between Keswick Dam (RM 302) and Red Bluff Diversion Dam (RM 243) throughout the year; however, the order provides the USBR with flexibility to contract the coldwater zone upstream in consultation with state and federal fishery agencies. The 1993 NMFS Biological Opinion requires the USBR to maintain average daily water temperatures at or below 56°F (13°C) from Keswick Dam to Bend Bridge (RM 260), though the BO also provides the USBR with flexibility to contract the coldwater zone upstream to Jelly's Ferry (RM 268) depending on a combination of water year type and Shasta Dam storage volume. In practice, the USBR generally operates the Shasta and Trinity divisions of the CVP to achieve the 56°F (13°C) water temperature standard between Bend Bridge (RM 260) and Jelly's Ferry (RM 268) each year, as determined by the Sacramento River Temperature Task Group.

The USBR recently proposed moving the water temperature compliance point upstream to Ball's Ferry (RM 276) (USBR 2004). The USBR cited recent escapement surveys conducted by CDFG which indicate the majority of winter-run salmon spawning in the Sacramento River occurs above Ball's Ferry in most years. Recent aerial redd surveys conducted by CDFG suggest that the vast majority of winter-run spawning occurs between Keswick Dam (RM 302) and Airport Road Bridge (RM 284) in most years (CDFG 2002, 2004). CDFG carcass surveys conducted since 1996 also suggest that there has been a recent upstream shift in the spatial distribution of winter-run spawning (Snider et al. 1998, 1999, 2000a, 2001).

The USBR suggests that moving the water temperature compliance point upstream to Ball's Ferry will provide dam operators with greater flexibility to manage the coldwater pool in Shasta Reservoir. This requirement springs from the California drought of 1976–1977, when Shasta

Reservoir storage was so low that only warm water could be released from Shasta Dam. As a consequence, the winter-run Chinook salmon population was decimated as adult and egg mortality increased from high water temperatures and concentrations of contaminants in the Sacramento River (Hallock and Fisher 1985, Hallock and Rectenwald 1990). Surface water temperatures in Shasta Reservoir increase through the summer and early fall from prolonged solar exposure and high ambient air temperatures. Consequently, the USBR must release water from the hypolimnion of Shasta Reservoir to achieve the 56°F (13°C) temperature target in the Sacramento River during the later phases of winter-run egg incubation, fry emergence, and fry dispersal. Hypolimnetic releases from Shasta reservoir drain the coldwater pool, thereby reducing the volume of coldwater that can be carried over for water temperature management in the following year.

Because the vast majority of winter-run redds are located upstream of Airport Road Bridge (RM 284) in most years, moving the water temperature compliance point upstream to Ball's Ferry (RM 276) would likely have little negative consequence for winter-run spawning and egg incubation. However, the contraction of the cold water zone in the Sacramento River could produce negative effects on winter-run holding and rearing habitat. Winter-run adults generally do not spawn immediately after entering their natal streams. Most winter-run adults migrate upstream between January and May, with peak migration in mid-March (Hallock and Fisher 1985), but spawning activity generally peaks in June. During the interval between migration and spawning, winter-run adults hold in pools to allow gonad development. High water temperatures can reduce the viability of eggs that are developing within female salmon as they hold. Winter-run salmon are generally less fecund than other runs of Chinook salmon that spawn in the Sacramento River, and these lower rates of fecundity may compromise the ability of the population to rebound from population crashes like that of the 1976 and 1977 spawning classes (Hallock and Fisher 1985). There is little information about where winter-run adults hold in the Sacramento River, so it is not clear if a contraction of the cold water zone would increase water temperatures in key areas of winter-run holding. It is likely that winter-run adults hold in habitats near where they spawn, which would suggest that winter-run adults generally hold in areas between Keswick Dam (RM 302) and a location near Airport Road Bridge (RM 284). Consequently, moving the temperature compliance point upstream to Ball's Ferry (RM 276) would likely have little effect on winter-run holding habitat and egg viability within holding females.

Moving the water temperature compliance point upstream could also affect winter-run fry rearing habitat, thereby influencing fry growth rates, and survival. The majority of winter-run fry begin dispersing downstream soon after emergence. As they grow, they become more tolerant of higher water temperatures, and if food resources are abundant, higher water temperatures can promote faster growth, which typically increases survival. However, higher water temperatures can also stress salmonid fry and make them vulnerable to other factors affecting health and survival, especially in the absence of an abundant food supply. There have been a few studies of invertebrate production (Stillwater Sciences 2003, USFWS 2005a, CFDG 1983), juvenile salmonid growth (Limm and Marchetti 2003), and salmonid rearing habitat in the Sacramento River (CDFG 1997, 1998, 1999, 2000; USFWS 2005b). However, it is not clear how the upstream movement of the water temperature compliance point would likely affect salmonid rearing habitat conditions in the upper Sacramento River. Rearing habitat conditions in the mainstem channel are especially important for winter-run salmon fry, because they rarely benefit from any floodplain or bypass flooding because of their emigration timing. In contrast, the juveniles of other salmonid runs periodically benefit from floodplain and bypass flooding, which can promote faster growth and higher survival (Sommer et al. 2001) and contribute to strong year classes.

Moving the water temperature compliance point upstream to Ball's Ferry (RM 276) could also affect winter-run fry survival by influencing the distribution and abundance of potential predators. Colder water temperatures can deter centrarchids from migrating farther upstream, and it can also depress predator feeding activity. Contraction of the cold water zone in the Sacramento River could allow predators to move farther upstream in higher densities, thereby increasing the predation exposure of winter-run fry. Little is known about the distribution and abundance of potential salmonid predators in the Sacramento River and how water temperatures can influence predator distribution, abundance, and feeding activity. Consequently, it is difficult to predict if movement of the water temperature compliance point would significantly affect the predation mortality of winter-run fry.

The 56°F (13°C) water temperature target in the Sacramento River is mandated by regulations designed to protect the endangered winter-run Chinook salmon. However, water temperature conditions also affect the other salmonid runs that spawn in the Sacramento River. We hypothesize that the upstream movement of the water temperature compliance point would have the greatest effect on the late-fall-run salmon population in the Sacramento River. The life history timing of late-fall-run salmon in the Sacramento River requires that a significant fraction of juveniles oversummer in the Sacramento River before emigrating downstream. Late-fall-run salmon spawning generally peaks in January, so fry emergence doesn't occur until April through June (Vogel and Marine 1991). By this time, water temperatures in the lower Sacramento River are usually too high to support salmonid fry survival, so juveniles rely on the cold water releases from Shasta to provide suitable rearing conditions throughout the summer. Contraction of the cold water zone in the Sacramento River would likely reduce the amount of summer rearing habitat available to late-fall-run juveniles. Chapter 4.5 explains that rearing habitat is most likely to be the limiting factor for late-fall-run Chinook salmon because of this oversummering life history strategy. Consequently, the loss of summer rearing habitat in the Sacramento River harbors the potential to reduce the late-fall-run salmon population.

Chapter 4.5 also describes our hypothesis that the location of late-fall-run spawning is influenced by the location of suitable water temperatures to support summer rearing of juveniles. Late-fall-run salmon fry and juveniles have limited swimming ability; consequently, it is unlikely that they migrate far upstream to pursue retreating cold water temperatures. We hypothesize that, over time, a selective pressure has been applied to the population so that the majority of late-fall-run spawning now occurs above RM 243 because summer water temperatures downstream of this point are unsuitable for juvenile rearing in the summer. Thus, moving the water temperature compliance point upstream to Ball's Ferry (RM 276) could reduce not only late-fall-run rearing habitat, but also spawning habitat.

Because a significant fraction of spring-run Chinook juveniles also oversummer in natal streams before emigrating, upstream contraction of the cold water zone could produce similar effects on the availability of both rearing and spawning habitat. However, it is unlikely that this would be a problem in the Sacramento River. The mainstem spawning population of spring-run Chinook salmon has been reduced dramatically, and it has hybridized with the fall-run Chinook salmon population because of hatchery practices and the loss of spatial segregation with fall-run salmon spawning. Consequently, it is questionable if a distinct, self-sustaining population of spring-run Chinook exists in the mainstem Sacramento River. Restoration of a distinct and viable population would also be contingent on restoring the spatial segregation with fall-run salmon spawning in the river. Secondly, spring-run salmon spawn during the warmest time of the year in the Sacramento River basin, which likely influences adults to spawn in locations where egg incubation and fry rearing can be supported. Upstream contraction of the cold water zone could thus reduce the amount of spawning habitat available to a restored population of spring-run salmon in the Sacramento River. However, it is likely that the vast majority of spawning would occur upstream of Ball's Ferry (RM 276), as occurs with the winter-run salmon population.

Upstream movement of the water temperature compliance point would likely have little effect on the fall-run salmon population in the Sacramento River. Adults generally spawn during a time of declining air and water temperatures, so that little risk would be posed to incubating eggs and emergent fry. Fall-run fry also emerge and disperse downstream before air and water temperatures increase dramatically in the late spring.

10.1.3 Implementation of the meander zone concept

A meander zone for the Sacramento River is an essential concept for improving ecological processes and habitats to benefit native species (SRCAF 2003). The fundamental processes of bank erosion and meander migration underlie the formation of numerous habitats that are critical for multiple species. As described in Chapter 7, bank erosion can create vertical cutbanks to support bank swallow nesting, and Chapters 4 and 10.1.1 describe the importance of bank erosion for recruiting sediment to the middle Sacramento River to build and maintain aquatic habitats (e.g., point bars). Chapter 3 describes how bank erosion drives the process of meander migration, which is essential for promoting channel cutoff and the formation of off-channel water bodies that can benefit western pond turtles and various aquatic species (Chapter 8), and influence the distribution, composition, and structure of riparian vegetation (Chapter 9) that supports native songbirds. The habitat needs of every focal species selected for this report intersects in some manner with the processes of bank erosion and meander migration.

Organizations and individuals have begun to assemble a meander zone in recent years by acquiring flood-prone lands from willing sellers within the floodway corridor. Some of the acquired lands have been actively restored by planting native vegetation.

The restoration of bank erosion and progressive channel migration will be essential to restoring or maintaining numerous native fish, avian, terrestrial vertebrate and plant species in the Sacramento River. Bank armoring is especially deleterious for bank swallow, a California threatened species. It will be difficult to recover this species without targeted removal of rip-rap, which is necessary to increase the availability of suitable nesting habitat. Analysis of metapopulation models suggests that removal of 3–20% of existing rip-rap could help establish an equilibrium population of bank swallows on the Sacramento River (Moffatt et al. 2005). However, significant recovery for bank swallows via removal of bank armor would probably only be realized if restoration activities were focused along banks that are likely to provide suitable nesting habitat (e.g., at sites with appropriate soil conditions) and the population is large enough to expand into the new habitats.

10.1.4 Flow management for habitat formation and maintenance

Our review and analysis of existing reports and data suggests that relatively modest changes in the managed flow regime of the Sacramento River, coupled with physical habitat restoration, can improve habitat extent, quality, and utilization for multiple species. For example, Limm and Marchetti (2003) have found that seasonally inundated habitat that occurs within the bankfull channel provides important rearing habitat for salmon fry, suggesting that spring flow operations can be tailored to inundate these habitats at the appropriate time to improve fry growth and survival. Their research also indicates that well-timed spring pulse flows that reconnect these seasonally inundated habitats with the mainstem channel can prevent juvenile salmonid stranding. Similarly, the USFWS has surveyed potential stranding sites for juvenile salmonids in the upper Sacramento River and the flows that connect and disconnect these sites with the mainstem channel (USFWS 2005). This information can be used to make relatively small changes in the timing and shape of discharge declines to benefit salmon. Research by TNC (2005) and CDWR (Morgan and Henderson 2005a, 2005b) is supporting the development of spring flow characteristics that can be used to promote riparian vegetation recruitment (Chapter 9). The linkages between flow, meander migration, bank erosion and bank swallow habitat (Chapter 8) and off-channel water bodies that western pond turtles use as habitat (Chapter 8) also support the notion that management of certain flow regime characteristics is important for formation and maintenance of habitat for many species. It may be possible to improve the frequency of fundamental fluvial geomorphic processes that shape and maintain aquatic, riparian, and floodplain habitats by changing the pattern in which flows are released from reservoirs to evacuate flood storage, without requiring the dedication of any additional water supply.

In addition to this State of the System report, the Sacramento River Ecological Flows Study includes several field investigations and modeling applications that will help to refine flow characteristics of ecological processes that help to create and maintain a variety of habitats in the Sacramento River. These flow estimates will be contained in the final report, which is scheduled for distribution in the fall of 2007. The goal of these studies is to inform various water management planning efforts so that clearer environmental flow targets are considered while developing operational plans.

10.1.5 Red Bluff Diversion Dam

The gates of Red Bluff Diversion Dam (RBDD) are raised between 15 September and 15 May each year to promote upstream passage of the endangered winter-run Chinook salmon and the threatened spring-run Chinook salmon. However, the period of gate closure may still clip the tail end of the upstream migration period for spring-run Chinook salmon, which historically extended into June and may currently extend into August (Cramer and Demko 1997). Though there is relatively less monitoring of spring-run salmon above RBDD, available evidence suggests that escapements are few and there has been little evidence of substantial spawning in the mainstem Sacramento River (CDFG 2004). Temporal overlap in spring-run and fall-run Chinook salmon spawning also make it difficult to distinguish the number of spring-run spawners and spawning locations in the upper river basin (CDFG 2004). Nevertheless, if efforts are made to restore a distinct, self-sustaining population of spring-run Chinook salmon to the upper Sacramento River mainstem channel and tributaries (e.g., Battle and Clear creeks), then current gate operations at RBDD have the potential to conflict with late upstream migrants of a threatened species.

RBDD gate operations also clip the latter portion of the upstream migration period of green sturgeon, which may extend into late July (Moyle 2002). The southern DPS of green sturgeon was recently listed as threatened by NMFS (2006), so RBDD gate operations are currently conflicting with another listed species. Little is known about the specific habitat requirements and locations of green sturgeon spawning in the Sacramento River, but most sturgeon are believed to spawn above RBDD unless passage is blocked. Relatively few spawners migrate upstream each year, so any reduction in the spawning population or the spawning success rate can negatively affect the population. Current radio telemetry studies may improve our understanding of the upstream migration patterns of green sturgeon, which will permit a more reasoned assessment of the risk that current RBDD gate operations pose to the southern DPS of green sturgeon.

10.2 Recommendations

Based on the conceptual models contained in the focal species chapters and the preceding synthesis section, we can recommend some restoration activities that will likely contribute to improving the health and status of the focal species in the Sacramento River. Though we are reasonably confident that these recommendations will improve habitat conditions for our selected focal species, each activity should be implemented as part of an adaptive management program whereby restoration actions are designed to test hypotheses and sufficient resources are dedicated for focused monitoring of system response.

10.2.1 Remove the coarse surface layer to expose subsurface gravels

Chapters 3 through 5 discuss the process of bed coarsening that has likely occurred in the upper Sacramento River, as high flow releases from Shasta Dam have recruited gravel stored in the channel bed until larger lag particles cover the surface and trap spawning-sized sediment in the subsurface. Nearly 250,000 yds^3 (190,000 m^3) of gravel has been introduced to the Sacramento River by state and federal agencies since 1979, and these gravel injections have likely played an important role in maintaining local patches of spawning habitat located near injection sites. However, the scale of gravel augmentation to date has likely done little to expand salmonid spawning habitat significantly in light of the 10 million yds^3 (7.6 million m^3) of coarse sediment mined from the basin or trapped by water supply dams.

One of the primary approaches for restoring spawning habitat in the Central Valley has been to bury armored channel beds with layers of gravel sufficiently deep to support spawning, or to stage spawning-sized gravel along channel margins and banks to be recruited to the channel by high flow events. These approaches ignore the significant volume of ecologically valuable sediment already stored in the subsurface. Considering the cumulative deficit of coarse sediment in the upper Sacramento River and the resources required to mine, process, transport, and inject gravel into the river, tapping the supply of coarse sediment stored in the channel subsurface may be a relatively cost-efficient and timely strategy for increasing spawning-habitat area. It may be possible to expose spawning-sized gravel stored in the channel subsurface of the upper Sacramento River by simply removing the coarse surface layer mechanically. This approach should be implemented first as pilot-scale investigations in which cobbles are removed by hand from the surface of small patches (e.g., 100 ft x 100 ft [30 m x 30 m]) to test the effects on spawning habitat utilization.

There are at least three methods for identifying zones where removal of the coarse surface layer could increase spawning habitat. The USFWS has conducted River2D modeling to simulate spawning conditions for several reaches of the upper and middle Sacramento River. The River2D model simulations can identify where hydraulic conditions (e.g., water depth and velocity) appear to be suitable to support spawning, but where the bed surface is too coarse. Such sites are candidates for removal of the coarse surface layer to expose finer material stored in the subsurface. A separate, but similar, method would involve using the professional judgment of experienced biologists to identify areas that seem to have suitable hydraulic conditions to support spawning, but where a coarse bed surface appears to prevent spawning. A third method would include removing the surface layer of zones that are adjacent to existing patches of spawning habitat. These methods are not mutually exclusive, and each method will likely need to be applied to cover the range of existing and potential spawning habitat that occurs in the upper Sacramento River.

Removal of the coarse surface layer will expose subsurface gravel to transport during successive high flow events, so this approach for expanding spawning habitat will work best when coordinated with a gravel augmentation program. Otherwise, scour of the exposed material could stimulate local channel incision.

10.2.2 Increase the frequency of gravel augmentation below Keswick Dam

Chapters 3 through 5 discuss the effects of dam construction and aggregate mining on the gravel supply and aquatic habitat in the upper and middle Sacramento River. As compared with historical conditions, the Sacramento River has been deprived of millions of cubic yards of coarse sediment since the construction of Shasta Dam in the mid-1940s. Habitat surveys have demonstrated the loss of spawning habitat caused by the reduced sediment supply and high flow events that have scoured gravel from the channel bed. State and federal agencies have added more than 242,000 yds^3 (185,000 m^3) of spawning-sized gravel to the upper Sacramento River since 1979, but this scale of gravel augmentation is small when compared with the cumulative deficit of coarse sediment supplied to the river over the past six decades. Based on Buer's estimate that the upper watershed provided an average annual gravel supply of approximately 50,000 yds^3 (38,000 m^3) the volume of augmented gravel only compensates for about five years of the historical sediment supply. Gravel injections implemented to date have likely played an important role in maintaining local patches of spawning habitat located near the injection sites, but they have done little to expand the cumulative spawning habitat in the upper Sacramento River.

In the early 1980s, CDWR and CDFG proposed a gravel augmentation program that would add 1 million yds^3 (0.8 million m^3) of spawning-sized gravel to the upper Sacramento River (CDWR 1980). Approximately 100,000 yds^3 (76,000 m^3) of sediment was added to the channel in 1990 and 1991 as part of the initial phase of this project (Bigelow 1996), but the program was never completed. The USBR initiated a separate phase of gravel augmentation in 1997, but the volume of added gravel was still short of the million cubic yards originally envisioned by CDWR and CDFG. In light of the cumulative deficit of coarse sediment since the construction of Shasta Dam, more frequent gravel augmentation may be required in the upper Sacramento River to expand salmonid spawning habitat. As described in Chapter 4, the life history strategies of both fall-run and winter-run Chinook salmon make them vulnerable to spawning habitat limitations, so any improvement in the extend and quality of spawning habitat will likely benefit those two runs most. As with previous gravel augmentation efforts, gravel injection should be focused between Keswick Dam (RM 302) and Clear Creek (RM 290) where there is little sediment supplied to the channel and where the bulk of salmonid spawning occurs. Since the improvement of fish passage facilities at ACID Dam (RM 298.4) in 2001, there has been an upstream shift in the distribution of winter-run salmon spawning (CDFG 2002, 2004). Other salmonids also exhibit a spawning preference for the reach above ACID Dam. However, the reach between Keswick Dam (RM 302) and ACID Dam (RM 298.4) has likely experienced some of the worst bed coarsening since Shasta Dam interrupted the upstream sediment supply. SCUBA surveys of the channel bed between Keswick and ACID dams indicate that much of the bed is covered by an armor layer (Bigelow 1996). The combination of a salmonid spawning preference above ACID Dam with pronounced bed coarsening in this reach creates a high risk of redd superimposition. Consequently, we recommend that initial gravel augmentation efforts be focused primarily in the reach between Keswick Dam (RM 302) and ACID Dam (RM 298.4).

While it is difficult to identify a target volume for a larger-scale augmentation program, it is not necessary to restore all of the gravel supply that has been lost to the system since the completion of Shasta Dam. Changes to the flow regime have reduced the frequency of bed mobilizing events so that much of the gravel stored in the channel bed has a longer residence time, as compared with historical conditions. Also, the ultimate goal of a gravel augmentation program is to increase aquatic habitat, primarily for salmonids, which can be achieved by restoring less than the full deficit of coarse sediment. Though the scale of the USBR and CDWR gravel augmentation projects implemented to date (~100,000 yds^3 [76,000 m^3]) seems to have been successful in maintaining local patches of existing spawning habitat, the infrequency of gravel augmentation has done little to compensate for the cumulative loss of material to the system. As part of the larger Sacramento River Ecological Flows project, Stillwater Sciences has developed a new sediment transport model (Cui, in press) to assess the effects of different restoration and management options, such as changes in the flow regime and sediment supply. These modeling applications will allow us to make more specific recommendations about the scale of gravel augmentation to implement in the upper Sacramento River. These recommendations will be included in the Final Report for the Sacramento River Ecological Flows Project, which is scheduled for distribution in the fall of 2007.

The initial purpose of a large-scale augmentation program is to compensate for the reduced sediment supply to the Sacramento River since the completion of Shasta Dam, but additional gravel injections will be required following future high flow events. The initial gravel augmentation program should occur over a short period of time (e.g., a few years) to jumpstart the process of re-building aquatic habitat by increasing coarse sediment storage in the channel.

Increasing the frequency of gravel augmentation in the upper Sacramento River raises the question of locating an appropriate gravel supply. Mining gravel from the floodplain or from tributaries may simply shift the location of the effects of a reduced gravel supply, or create other ecological impacts (e.g., floodplain pits). CDWR explored the possibility of mining sediment from depositional zones of Shasta Reservoir, which can be exposed during periods of low storage (Buer 1994a). Although the material mined from the reservoir would be a “free” or low-cost resource for state and federal agencies, the actual mining, processing, and transporting of coarse sediment would likely be logistically difficult and costly, especially in the steep terrain surrounding Shasta Reservoir. Nevertheless, we recommend that state and federal agencies conduct a feasibility study of using sediment from Shasta Reservoir as a long-term source of material for the upper Sacramento River. A mining program would also have the added benefit of prolonging Shasta Reservoir storage. Another potential source of material may be dredger tailings that cover acres of BLM land in the Clear Creek drainage (Buer 1994a), though this material may be needed for planned ecosystem restoration in Clear Creek. Another potential source of coarse sediment may be in-stream gravel sinks that occur in the vicinity of Redding. Chapter 3 describes how some current river features, such as Kutras Park, are the remnants of the massive aggregate mining operations that supported the construction of Shasta Dam and related infrastructure. Several of these mining pits are located in or near the mainstem channel, so they may capture gravel that routes as bedload during high flow events. Mining coarse sediment from these pits may provide a relatively local and low-cost source of gravel for salmonid habitat restoration below Keswick Dam.

As described above, we expect to make more detailed recommendations about a target volume for a gravel augmentation program as part of the Final Report for the Sacramento River Ecological Flows Project. In the meantime, we recommend that state and federal agencies begin the process of identifying potential sources of gravel for a large-scale augmentation program in the

Sacramento River. The investigation should update the work completed by CDWR (1994) to identify gravel sources for ecological restoration, and it should initiate more detailed planning-level analyses for the most promising gravel sources.

10.2.3 Use ACID Dam to redistribute salmonid spawning

Keswick Dam (RM 302) defines the upstream limit of anadromy on the mainstem Sacramento River, and several of the mainstem spawning populations of salmonids concentrate spawning in the 10 mi (16 km) reach downstream of the dam (Snider et al. 1998, 1999, 2000a, 2000b, 2000c, 2001). As discussed in Chapter 4, a comparison of available spawning habitat in the upper Sacramento River with the number of redds recorded by CDFG during periodic aerial redd surveys suggests that redd superimposition may be occurring in the Upper Sacramento River. However, it is unclear if redd superimposition is occurring to a degree that is affecting any of the mainstem spawning populations. Redd superimposition can occur in a stream with abundant habitat. Adult salmon that arrive later in the spawning period may be predisposed to construct redds atop existing redds because of superior intragravel flow created by previous redd construction, which can loosen the gravel and can clean fine sediment from the patch. So documenting the occurrence of redd superimposition is not as important as documenting the scale at which it occurs, to determine if it has a population-level effect. The risk of redd superimposition is usually greatest for those spawning populations with relatively high escapements (e.g., fall-run Chinook salmon) as greater numbers of adults compete for limited habitat resources. However, salmonid populations with relatively low escapements (e.g., winter-run Chinook salmon) can be vulnerable to redd superimposition as well, because spawning sites are not uniformly distributed among all available habitats. For example, the distribution of winter-run Chinook salmon spawning indicates that the population does not utilize spawning areas where hydraulic conditions, bed material, and water temperatures would seem to support spawning. Rather, adults bypass suitable spawning areas during their upstream migration, concentrating redds in the 10-mi (16-km) reach below Keswick Dam.

There may be several factors that compel winter-run Chinook salmon to bypass apparently suitable spawning habitat. Adults may be genetically hard-wired to travel as far upstream as possible to spawn, reflecting the historical migration of winter-run into the upper elevations of the drainage prior to the construction of Shasta Dam. Environmental conditions in the past 30 years may also have influenced the current distribution of spawners, because elevated water temperatures during dry water years may have eliminated the progeny of adults that once spawned farther downstream. As a result, the current population of winter-run Chinook salmon may be descended from adults that spawned closer to Keswick Dam where water temperatures were relatively cooler during periods of environmental stress.

Even if salmon are genetically predisposed to spawn in the most upstream reaches of the mainstem Sacramento River, it may be possible to reduce redd superimposition by modifying and operating the fish passage facilities at ACID Dam (RM 298.5) to influence the distribution of spawning. In 2001, fish passage was improved at ACID Dam (RM 298.5), and there was an immediate upstream shift in the distribution of winter-run Chinook salmon spawning (CDFG 2002, 2004).[4] This upstream shift highlights the potential for operating the ACID Dam fish

---

[4] This upstream shift in spawning may have actually increased the risk of redd superimposition for winter-run as more adults bypassed unused but suitable spawning habitat below ACID Dam to construct redds in the relatively limited spawning habitat above ACID Dam.

passage facilities to reduce the risk and effects of redd superimposition. Once spawning habitat above ACID Dam is saturated with redds, the fish passage facilities at the dam could be used to impede upstream passage and compel successive spawners to utilize the habitat downstream of the dam, thereby protecting the constructed redds upstream. ACID Dam provides one of the few opportunities for using an existing barrier to influence the distribution of spawning, because navigation and recreational uses of the river would complicate the use of channel-spanning fish racks farther downstream.

The number of salmonid redds that can be accommodated in the habitat above ACID Dam will vary with time as gravel augmentation activities restore some of the lost gravel supply and as high flow events scour and re-deposit sediment in the reach. As part of field study for the Sacramento River Ecological Flows Project, we have recorded 2005 fall-run redd locations using helicopter videography, which will be used in conjunction with recent aerial photos and a GIS to estimate the amount and location of salmonid spawning habitat in the upper Sacramento River. This analysis will provide an estimate of the number of salmon redds that can be supported in the reach above ACID Dam (RM 298.5), which can provide a general metric for determining when to block upstream migration to compel spawning downstream of ACID Dam. However, high flow events during the 2006 water year may have already altered salmonid spawning habitat extent and distribution within the Sacramento River, which highlights the need to periodically re-calculate spawning habitat carrying capacity as conditions change.

Researchers in the Sacramento River have observed redd superimposition in the field; however, the scale of redd superimposition is not clear, so it is not possible to assess if redd superimposition is having any sort of population level effect for any of the salmonid populations that spawn in the upper Sacramento River. Our analyses suggest that redd superimposition likely causes significant mortality for multiple runs of Chinook salmon, but there has been no field-based documentation of the scale of redd superimposition. Consequently, any consideration of modifying and operating the fish passage facilities at ACID Dam to influence the distribution of spawning should be preceded by a redd superimposition study that not only tests whether redd superimposition occurs, but also includes population modeling to assess the significance of mortality caused by any observed redd superimposition.

We should also caveat that we have not researched the requirements of using ACID fish passage facilities as a migration barrier, nor have we discussed this idea with ACID. Any use of ACID facilities to manage salmonids in the Sacramento River requires the willing participation of ACID. Our goal here is to introduce a potential restoration measure that could benefit salmonids in the Sacramento River. Further study is required if this restoration approach is deemed to have merit.

10.2.4 Avoid future bank armoring unless comprehensive study of impacts is conducted and appropriate mitigation planning is implemented

Bank armoring alters ecological processes and can adversely impact many species, both terrestrial and aquatic (see 10.2.5 below). A species that is particularly sensitive to bank armoring is the state-listed bank swallow. Given the historical habitat losses and reduction in the Sacramento River bank swallow population, no new bank revetment projects should be conducted on the Middle Sacramento River without thorough assessment of their potential short- and long-term effects on bank swallows and identification of appropriate mitigation (avoidance, minimization, and compensation) of adverse impacts. When avoidance or minimization approaches are not feasible, compensatory mitigation actions should be focused on removal or, in appropriate locations, abandonment (i.e., stop maintaining certain sections of revetment that are being actively eroded by the river if they are no longer needed) of existing riprap or setting back levees (see Section 10.2.5) in areas that (1) are not critical for protection of vital human infrastructure, (2) contain suitable soils and channel migration potential for creation of bank swallow nesting habitat, and (3) have landowners willing to cooperate. Mitigation ratios for habitat loss of greater than 1:1 would help contribute towards recovery of the bank swallow population. Creation of a mitigation bank should be explored as a potential strategy that could help maximize short- and long-term benefits to bank swallows by creating large areas in which natural processes provide a dynamic landscape with a reliable supply of suitable nesting habitat.

Mitigation for bank armoring might include consideration of levee removal at select sites, which may help promote progressive meander migration in a way that expands usable habitat for bank swallows. This can lead to immediate benefits for bank swallow populations, as demonstrated on the Sacramento River, after a levee removal and rip-rap retirement project was completed at RM 233 in late fall 1999 (Golet et al. 2003). Erosion induced by winter storms expanded an existing cut bank, and a swallow colony from nearby established itself there in the spring of 2000. The newly established colony, with 2,770 borrows, was the largest on the river that year.
It represented a substantial expansion for bank swallows at the site, which had supported just 930 burrows in the previous year. While this single anecdotal account may not be entirely representative of potential gains at other sites, it does suggest that levee setback is a viable option for creating new bank swallow habitat if locations are chosen wisely. Levee setbacks to promote meander migration and cutoff processes would also benefit many other species, including western pond turtle and Fremont cottonwoods, as discussed in Chapters 8 and 9.

Management that focuses on increasing erodibility can be site specific, unlike management that focuses on changes in flow. Under ideal conditions, sections would be chosen such that increases in erodibility and area connected to the mainstem will lead to direct benefits for key species and habitats. For example, an ideal location for riprap removal would be a bank that is composed of materials that are suitable for bank swallow nesting (as discussed above), and that has a suitable point bar for riparian recruitment on its opposing, inside bend. Expanding the area available for meander migration should also lead to channel cutoffs and creation of off-channel aquatic habitats suitable for western pond turtle and initiation of successional processes that create a diverse mosaic of habitat types, including recruitment of new cottonwood and mixed riparian forests.

Ideal locations and designs for removal of bank armor can be assessed by running a series of scenarios through a meander migration model (Larsen et al. 2006). The results of such a model would be a key input for incorporation into the SacEFT which help in evaluating the relative benefits of proposed actions in terms of quantifiable metrics such as average annual migration rates and the amount of floodplain reworked. The SacEFT should also help account for how changes in erodibility are likely to interact with any management-related changes in flow releases.

10.2.5 Setback levees to expand the meander migration zone

In order to maintain or enhance the potential for future cutoff formation through channel migration, opportunities for land-based management activities such as setting back levees or removing rip-rap (see Section 10.2.4 above) to increase the functional meander migration zone should be considered (Section 10.1.3). These can be highly controversial discussions, however, stakeholders, managers, and resource agencies have demonstrated an ability to work together

Sacramento River to find locations where actions such as these may provide multiple benefits (e.g., to improve ecosystem health and flood protection, Golet et al. 2006).

One way to inform selection of levee setback sites and designs is through the use of meander migration modeling (Larsen et al. 2006). In one recent study, the effects of alternative setback scenarios were assessed for a 17 mile-long (28 kilometer-long) reach below Pine Creek (RM 196) (Larsen et al. 2006). Simulations showed that migration rates could be increased by nearly twofold for even the least ambitious (i.e., 330 ft [100 m]) setback scenarios. In segments that have suitable soils and bank heights, this could greatly increase habitat for bank swallows. For more ambitious (i.e., 2,600 ft [800 m]) scenarios, migration was accelerated by eightfold, and was shown to produce conditions that were generally favorable for cutoff—which would produce the off-channel aquatic habitats required by other species, including the western pond turtle (see Chapter 8), and provide potential establishment sites for Fremont cottonwood and other riparian plant species (see Chapter 9). Intermediate setback distances yielded intermediate migration rates and created conditions that were somewhat less favorable for cutoff, relative to the 2,600 ft (800 m) setback option. Taken together, these results suggest that a range of outcomes are possible, and that a variety of species can benefit from management via levee setback. Selecting one option over another will be an issue of balancing the estimated benefits for each individual species of concern with potential costs.

The output of the meander migration model (Larsen et al. 2006) can be used as key input for incorporation into the SacEFT which helps in evaluating the relative benefits of proposed actions in terms of quantifiable metrics such as average annual migration rates and the amount of floodplain reworked.

10.2.6 Require gravel augmentation as mitigation for bank armoring

Mitigation for bank armoring often focuses on compensating for the loss of cutbanks, shaded riverine aquatic (SRA) habitat (Fris and DeHaven 1993), or LWD recruitment to the channel (DeHaven 2000). However, CDWR has demonstrated that bank armoring has a significant impact on riverine habitats, including point bars located downstream of protected banks (Buer 1994b). In the upper Sacramento River between Keswick Dam (RM 302) and RBDD (RM 243.5), *tributaries* supply approximately 85% of the gravel (California Resources Agency 1989). However, in the middle Sacramento River, *eroding banks* supply approximately 85% of spawning-sized gravel (Buer 1984). The loss of this gravel supply from bank armoring causes downstream point bars to be smaller than those downstream of naturally eroding banks (Buer 1994b), which reduces salmonid spawning and rearing habitat. Salmon often spawn at the head of point bars, so a reduction in the size of a point bar likely reduces the prime spawning habitat that salmon prefer. The loss of spawning habitat by reduced point bar size likely affects only fall-run Chinook salmon, because it is the only run that spawns in the middle Sacramento River in significant numbers. In contrast, the loss of rearing habitat associated with reduced point bars can affect all of the salmonid runs, because all juvenile salmonids migrate through the middle Sacramento River on their route to the ocean. Smaller point bars reduce salmonid rearing habitat by reducing the area of the eddy zones that form on the downstream size of point bars. These eddy zones are usually important rearing habitats because the low flow velocities allow juvenile salmonids to minimize energy expenditure to maintain position, while adjacent high velocity water delivers insect drift to promote growth (Stillwater Sciences et al. 2003).

The downstream ends of point bars often provide favorable conditions to promote cottonwood seedling colonization because of fine sediment deposition and a more gradual decline of the wetted surface as compared with the upstream ends (TNC 2003a, Wood 2003a, Morgan 2005; see also Section 9.2). Thus, a reduction in the size of point bars can also reduce the surface area available to support riparian vegetation recruitment.

Because of the importance of eroding banks as sources of gravel for the middle Sacramento River, any future bank armoring activities should mitigate not only for the loss of SRA and LWD, but also for the lost recruitment of gravel to the channel. CDWR has developed field-based methods for estimating the gravel stored in an eroding bank (Buer 1994b), which can be utilized to help determine gravel mitigation requirements for specific bank armoring projects.

Though gravel augmentation would compensate for some of the lost sediment supply caused by bank armoring, it will not address many of the other effects on aquatic, riparian, cutbank, and off-channel habitats. Even if added gravel helps to maintain point bars downstream of armored banks, the revetment still reduces salmonid habitat in the vicinity of the armored bank by causing deeper scouring of pools and a steeper cross sectional geometry (Buer 1994b). Similarly, bank armoring prevents the formation of vertical cutbanks that can provide potential nesting habitat for bank swallows if the bank erosion occurs in appropriately textured soils. The prevention of meander migration also reduces the potential for channel cutoff, which is essential for creating new off-channel water bodies (e.g., oxbows) that support a multitude of fish, avian, and amphibian species.

10.2.7 Release spring pulse flows to inundate shallow water habitats within the bankfull channel

Chapter 4 discusses the importance of shallow water habitat in promoting fast growth and survival of juvenile salmon. Previous research in the Sacramento River indicates that inundation of the flood bypasses that flank the Sacramento River enhances growth and survival of juvenile salmonids (Sommer et al. 2001), but bypass flooding is contingent on infrequent high flow events that are usually tied to flood management operations. In contrast, the inundation of shallow water areas within the bankfull channel can occur much more frequently as a function of normal water operations because lower flow magnitudes are required. Previous research suggests that inundation of these habitats may provide important rearing habitat for juvenile salmonids in the Sacramento River (Maslin et al. 1997), but stranding may occur in the late spring as these marginal habitats become disconnected from the mainstem channel (Limm and Marchetti 2003). Shallow-water rearing habitats within the bankfull channel can be especially important for the winter-run Chinook salmon population. Winter-run fry emerge and disperse downstream primarily between August and November when there is relatively little floodplain or bypass inundation, so the population does not benefit from the periodic expansion of shallow water rearing habitat that can contribute to strong year classes of other salmonids.

We recommend that flows be released to inundate shallow water habitats within the bankfull channel during periods of fall-run (February and March) and winter-run (September and October) fry emergence and dispersal. Fall-run and winter-run fry are the targets because their life history strategies require good fry production to maintain the health of the population, and fry typically rely on shallow water habitats for rearing. Inundating shallow-water habitat for winter-run salmon fry may be problematic, because it occurs at a time when the USBR is managing the coldwater storage pool of Shasta Reservoir to ensure adequate carryover storage for the following year. This strategy is also problematic for winter-run salmon because potential predators are likely more active during the period of winter-run fry emigration, as compared with that of fall-run population due to warmer water temperatures.

Because of the potential problems associated with inundating shallow water habitats for winter-run salmon fry, initial flow experiments should focus first on providing habitat for fall-run salmon fry in February and March. The field components that accompany this SOS Report will include an inventory of shallow water rearing sites in the middle Sacramento River and an estimation of the flows that support ingress and egress. This study will provide a better understanding of the flow magnitudes required to inundate these habitats and connect them with the mainstem channel to promote growth.

Previous research in the Sacramento River also indicates that salmon fry can become stranded in shallow water and marginal habitats when flow magnitudes recede (Limm and Marchetti 2003). Subsequent flow increases can temporarily re-connect these seasonally inundated and marginal habitats, allowing stranded fry to escape to the mainstem channel. The USFWS is conducting an inventory of potential stranding sites in the upper Sacramento River between Keswick Dam (RM 302) and Battle Creek (RM 271), including an estimation of the flows that connect and disconnect these marginal habitats with the mainstem channel in the upper Sacramento River (USFWS 2005b). Similarly, the inventory of shallow water habitats that will be conducted as part of the Sacramento River Ecological Flows Study will provide estimates of the flow magnitudes that connect seasonally inundated habitats with the mainstem channel in the middle Sacramento River. These two studies should provide water planners with a better understanding of how changes in spring flow magnitudes can potentially strand or rescue salmon fry from seasonally inundated habitat within the bankfull channel. The final report for the Sacramento Ecological Flow Study will contain the results of these field studies.

10.2.8 Manage recession flows to promote riparian seedling establishment

Recruitment of Fremont cottonwood seedlings is critical for sustaining riparian forests along the Sacramento River, but changes to the natural flow regime due to flow regulation can reduce the successful establishment of this species. Recent studies of cottonwood recruitment along the middle Sacramento River conducted by TNC (Roberts et al 2002, TNC 2003) and CDWR (Morgan and Henderson 2005a,b; Morgan 2005) have provided critical local information on three key factors known to limit cottonwood recruitment in other lowland alluvial river systems: site hydrology, seed release timing, and seedling water stress thresholds. This information has allowed TNC and CDWR to calibrate an existing conceptual model of cottonwood seedling recruitment for application to the Sacramento River (see Section 9.8.1 of the cottonwood chapter for more details). Managed flow releases have already been successfully applied for restoration of riparian trees elsewhere (e.g., Rood and Mahoney 2000, Rood et al. 2003, 2005).

The Sacramento-specific studies conducted by TNC and CDWR provide sufficient information to develop an experimental spring flow release to promote cottonwood recruitment in the middle river. Concurrent monitoring of seedling initiation and establishment could be used to track the success of such an experiment flow release, and fine-tune the details for subsequent application. A similar program is already underway on the Trinity River where high water conditions in spring 2006 allowed operators to release an experimental flow intended to promote cottonwood seeding recruitment at a specific restoration site. Monitoring currently underway should be highly informative for planning similar efforts in the Sacramento River system.

The volume of water available for a recruitment flow (and therefore the range of potential magnitude, duration, and flow recession) will be largely determined by contemporary hydrologic conditions. The ideal condition for promoting tree recruitment is to release a managed flow in a wet water year when upstream reservoirs are fairly full (from previous wet or above-normal years) (see Section 9.8.1 for more details). Under these conditions the flow pulse can be sustained to allow moist conditions to persist at fairly high relative elevations on floodplains until seedlings can grow extensive root systems and reach the perennial water table. Furthermore, recruitment flows are not necessary in every wet or above-normal year, but instead need only occur once every 5 to 10 years on average (so, in 25–50% of wet and above-normal years). Evidence from many alluvial river systems in the western United States suggests that this frequency of successful recruitment events is typically sufficient to maintain a healthy and dynamic cottonwood riparian forest ecosystem (Braatne et al. 1996, Rood et al. 2003 and 2005). Cottonwood recruitment tends to be highest when a successful recruitment flow year is followed by one or more normal flow years, which allow seedlings to survive after the first summer and become established by limiting mortality due to scour or inundation caused by high winter flows and by desiccation during summer low flow periods (McBain and Trush 2002, Stillwater Sciences 2006).

The results of the TNC and CDWR studies, coupled with results of similar studies in the San Joaquin Basin and elsewhere, indicate that cottonwood seedlings along the Sacramento River are most likely to establish in a band that is 3.3 to 6.5 ft (1 to 2 m) above the average stage during summer baseflow periods (approximately 8,500 cfs for the study sites evaluated) (TNC 2003, Morgan and Henderson 2005a). This means that a targeted recruitment flow should be in the range of 22,000 to 37,000 cfs during the peak of the cottonwood seed release period (roughly the last week of April through the first week of June in most years) (TNC 2003). A gradual stage recession in the range of 0.4 to 0.8 in/day (1 to 2 cm/day) (computed as a 3-day running average) is required to allow seedlings to survive by keeping their roots in contact with the receding soil moisture level. This translates to a flow recession rate of about 400–450 cfs/day until the flow drops to about 11,000 cfs, at which point the flow recession rate would probably need to drop to about 150 cfs/day until the late summer baseflow level is reached. For example, based on information contained in the TNC (2003) study, changes in the timing and recession rate of spring 2003 conditions could have transformed flows into successful recruitment flows without placing any new water demands on the system (i.e., the total water volume required would be equal to or less than the actual wet year flows at the study sites).

Regulated flow releases for cottonwood recruitment have a strong potential to benefit other species that are adapted to the large, regular disturbance imposed by the spring snowmelt pulse (Lytle and Poff 2004). For example, winter-run Chinook salmon (*Oncorhynchus tshawytscha*) smolts migrate out of the rivers to the ocean during this event (Section 4.2). The Vernalis Adaptive Management Program on the San Joaquin River is one restoration program that mimics elements of the spring snowmelt pulse to increase outmigrant success (SJRGA 2005). Similarly, managed fall releases to signal upstream migration timing (‘attraction flows’) are currently implemented every year. These flows may be designed to benefit riparian recruitment if the flow magnitude is sufficient to scour vegetation from potential seedbeds. In these two examples (spring and fall pulse flows), multiple species will benefit from flow restoration measures that are properly coordinated. As currently implemented, VAMP flows occur approximately a month too early to benefit cottonwood and willow species, and the ramping rates are generally too steep to sustain newly-germinated seedlings. Fall attraction flows are currently too modest to scour vegetation or deposit sediment on floodplains.

However, it is also possible that spring recruitment flows could have some negative impacts on other target species. For example, the bank swallow nesting period coincides with the cottonwood recruitment period. Elevated flows in the 22,000 to 37,000 cfs range designed to promote cottonwood recruitment on suitable seedbed elevations 3 to 6 ft (1 to 2 m) above summer baseflow levels (see TNC 2003 for details) might promote some localized episodes of bank erosions that could affect nesting success of some bank swallow pairs (see Chapter 7 for more discussion of this type of effect). Although the risk of significant adverse impacts appears small, monitoring of bank swallow colonies during experimental recruitment flows should be conducted and adaptive management used to refine recruitment flow hydrographs to maximize benefits and minimize negative impacts.

10.3 Suggested Studies

The focal species chapters in this report highlight the lack of basic information that complicates the management and restoration of the Sacramento River corridor. For many species, we lack critical information about:

the habitat preferences and tolerances of different life history stages;

the forces that influence the extent, distribution, and quality of required habitats; and

the trajectory of key habitat types in the Sacramento River corridor.

This section identifies several studies to address some of the basic information gaps identified in the focal species chapters, and it also defines studies to test key hypotheses that spring from the conceptual models contained in the focal species chapters. In addition to the studies recommended here, the continuation of basic status and trends monitoring in the Sacramento River (e.g., salmon carcass surveys, aerial redd surveys, RBDD rotary screw trapping, bank swallow surveys, bank erosion monitoring) will be important to improve our understanding of process-habitat-biotic linkages in the Sacramento River and system response to management interventions.

10.3.1 Correlate soil mapping with expected bank erosion to prioritize locations for potential bank swallow colonies

Chapter 7 describes the importance of bank erosion in the Sacramento River corridor to bank swallows, which require fresh vertical cutbanks in appropriately textured soils for nesting colonies. Because bank swallow colonies are only found in freshly eroding banks (see Chapter 7), bank armoring activities pose a direct conflict with bank swallow recovery. Bank swallows are currently listed as a threatened species under the CESA; any further reductions in the population could necessitate listing the species as endangered under the CESA or require federal listing and protection, thereby increasing the potential for future conflict with management activities.

To facilitate the recovery of Sacramento River bank swallows, and to reduce conflict with future bank armoring activities, it is essential to identify locations in the Sacramento River corridor where expected bank erosion will occur in conjunction with appropriately textured soils. Identifying potential nesting sites can guide the acquisition of floodplain parcels and easements from willing sellers, and it can facilitate the assessment of proposed bank armoring activities to avoid or reduce impacts on bank swallows. In addition, opportunities to retire riprap where it is nonessential for protecting infrastructure should be explored, particularly where existing rip-rap covers banks and soils known or expected to be suitable as bank swallow nesting habitat.

Additional survey and modeling studies of bank swallows along the Middle Sacramento River could improve our ability to manage the system to promote enhancement and recovery of the bank swallow population. Some actions to consider include:

- Continuation of the annual CDFG surveys of colonies along the Sacramento River from Red Bluff to Colusa (RM 243–143) is crucial. However, as a potential modification to the current methodology, researchers should consider increasing the frequency of surveys in the Redding to Red Bluff (RM 292–243) and Colusa to Verona (RM 143–81) reaches. This would help eliminate the small but nevertheless potentially significant survey data gap (see discussion in Chapter 7). Surveys for RM 292–243 and RM 143–81 would ideally be conducted every year, but if resources are limited, surveys in alternate years may suffice. Initial results from two or more consecutive years of surveys for the reaches in question might help shed light on an acceptable frequency for future monitoring.
- Linking the existing model of progressive meander migration (Larsen et al. 2002), and possibly a model of chute cutoff processes, to models of biotic responses (i.e., metapopulation models similar to that of Moffatt et al. 2005 or a more complex model if sufficient data are available for parameterization) to predict the effects of future management actions on bank swallow habitat and population response. Ultimately, such linked process-habitat-biotic response models should be used to refine the population viability analysis conducted by CDFG (1992) to improve estimates of population size and colony distribution needed to promote recovery and maintain a viable population.

10.3.2 Conduct water temperature modeling to assess the effects of moving the water temperature compliance point upstream

Section 10.1.2 explores the implications of moving the water temperature compliance point for the maintenance of average daily temperatures at 56°F (13°C) in the Sacramento River, as proposed by the USBR (2004). The discussion suggests that even though a contraction of the cold water zone will likely have little effect on winter-run spawning and egg incubation, the change could negatively affect the amount of spawning and rearing habitat available for the late-fall-run salmon population. We hypothesize that the location of late-fall-run spawning above RM 243 is governed by the downstream limit of where suitable summer rearing temperatures have been maintained in the past. Testing this hypothesis is important to understand the potential residual effects of moving the water temperature compliance point, and the potential for affecting a species of concern.

The USBR has developed a new water temperature model that can predict water temperature conditions by river mile under a range of meteorological and operational scenarios. The model has the potential to predict water temperatures between Keswick Dam (RM 302) and Knights Landing (RM 89) (R. Yaworsky, personal communication, 25 April 2005). We recommend that the USBR apply the water temperature model to recreate historical water temperature conditions for a range of years for which aerial redd surveys of late-fall-run salmon spawning exist. Correlating modeled water temperature conditions with the location of late-fall-run salmon spawning locations can provide an initial test of our hypothesis.

We also recommend that the USBR apply the water temperature model to future water operation scenarios under a range of meteorological conditions to assess how a change in the water temperature compliance point would affect summer rearing conditions for salmonids. This round of modeling could test the downstream extent of 64°F (18°C) under a range of flow and meteorological conditions during the summer rearing months. Similarly, the USBR should model water temperatures under a range of flow and meteorological conditions to determine if water temperatures can be maintained below 63°F (17°C) at Hamilton City (RM 200) during the summer months when larval green sturgeon are in the river.

10.3.3 Study Overwintering habitat/velocity refugia for juvenile steelhead

The conceptual model described in Chapter 5 suggests that over-wintering habitat for juvenile steelhead may be a limiting factor in the Sacramento River, because juvenile steelhead require velocity refugia to avoid displacement during high flow events. Juveniles that are swept downstream may not have the swimming ability to migrate long distances back upstream. Consequently, juveniles displaced downstream would likely experience summer water temperatures that are not suitable, and they may encounter higher densities of predators that feed more actively because of warmer water temperatures in downstream reaches.

Juvenile steelhead seem to avoid low-velocity, shallow water habitat along channel margins as velocity refugia because such habitats provide little cover from predators. Instead, steelhead typically use the interstices among coarse cobble, boulders, and rip-rap during high flow events (Bjornn 1971, Hartman 1965, Bustard and Narver 1975, Swales et al. 1986, Everest et al. 1986). We hypothesize that there is little velocity refugia for juvenile steelhead in the upper Sacramento River. To test this hypothesis, we recommend that a survey of steelhead overwintering habitat be conducted, focusing on patches of coarse sediment particles (e.g., armored banks and bridge pilings, cobble-bedded reaches) with little fine sediment. The survey will support an estimation of the winter carrying capacity of juvenile steelhead in water years with high flow events.

10.3.4 Conduct a redd superimposition study

As discussed in Chapters 4 and 10.2.3, we suspect that a significant amount of redd superimposition occurs in the upper Sacramento River, and that redd superimposition is having population-level effects for both the winter-run and fall-run salmon populations. However, it is important to document redd superimposition in the field and to develop an estimate of the intensity with which it occurs. Understanding the occurrence and scale of redd superimposition can facilitate an understanding of the significance of associated egg mortality, which can assist the process of identifying appropriate restoration actions, such as removing the coarse surface layer from reaches of channel, increasing the scale of gravel augmentation, and potentially operating ACID as a spawning barrier.

We recommend that a redd superimposition study be conducted first between Keswick Dam (RM 302) and ACID Dam (RM 298.4) because a significant percentage of each salmonid run spawns in this reach. Also, there has been a recent upstream shift in the distribution of winter-run salmon spawning following the completion of improved passage facilities at ACID Dam in 2001. The relative scarcity of spawning habitat above ACID Dam makes us concerned that the upstream shift in winter-run spawning may be exacerbating redd superimposition and associated egg mortality. As described in Chapter 4.2, the life history strategy of winter-run Chinook salmon requires relatively high rates of fry production, so the population is vulnerable to any increase in egg mortality caused by redd superimposition.

Though it is critical to study redd superimposition between Keswick Dam (RM 302) and ACID Dam (RM 298.5), we also recommend that a study be conducted between ACID Dam (RM 298.5) and the confluence with Clear Creek (RM 290), because of the density of spawning that occurs in this reach for several of the salmon runs (Snider et al. 2000b, 2000c, 2001).

10.3.5 Manipulate off-channel water bodies to study rates of terrestrialization

There are a number of divergent forces that may affect off-channel habitat formation and terrestrialization. Conversion of riparian habitat to agriculture may have increased cutoff by removing vegetation that stabilizes banks. Bank armoring has certainly reduced cutoffs by preventing bank erosion and progressive meander migration. Flood management may have reduced the frequency of large flow events that drive bank erosion and meander migration, but increased summer baseflows may have the opposite effect. Agricultural conversion and timber harvesting may have increased fine sediment loads to increase the rate of terrestrialization, but reduced periodicity of flows to inundate off-channel habitats may counteract these effects. All of these competing forces make it difficult to understand how constant changes in land use, flow management, and sediment loading interact to affect the rates of formation and senescence of off-channel water bodies.

The formation and terrestrialization of off-channel water bodies in the Sacramento River has been the focus of much recent study (Micheli and Larsen, in prep., Morken and Kondolf 2003). We are also conducting a field study that is part of the Sacramento River Ecological Flows Study that is designed specifically to explore terrestrialization rates of off-channel water bodies by examining sedimentation rates.

One potential measure for prolonging the lifespan of existing off-channel water bodies is to physically manipulate them by dredging them and by altering the manner in which they connect with the mainstem channel. Many off-channel water bodies are seasonally choked with aquatic vegetation, especially *Ludwigia*, which can increase the rate of terrstrialization by the addition of organic detritus (Levrat, pers. comm., 2005). Dense aquatic vegetation also affects the habitat value of off-channel water bodies for different species by influencing such variables as cover and dissolved oxygen. We hypothesize that water depth can control the establishment and cover of Ludwigia in an off-channel water body, and recommend that a set of off-channel water bodies be dredged to varying depths, with associated monitoring, to assess the effects on aquatic vegetation establishment, water chemistry and wildlife use patterns.

This Study can be conducted in conjunction with the western pond turtle survey and the riparian vegetation monitoring described in Section 10.3.7 and Section 10.3.8, respectively.

10.3.6 Increase green sturgeon research

NMFS recently listed the southern DPS of green sturgeon as a threatened species (2006), yet little is known about green sturgeon migration and habitat use in the Sacramento River. Researchers have initiated promising radio telemetry studies that may provide valuable information about green sturgeon behavior in the Sacramento River. Similarly, UC Davis researchers have conducted valuable laboratory studies of green sturgeon eggs and larvae using Klamath River stock, which can yield valuable information about habitat tolerance and the effects of different environmental conditions on growth rates. However, additional research is needed to facilitate the identification of appropriate conservation and restoration measures that will benefit the species in the Sacramento River.

Hundreds of adult green sturgeon captured in San Pablo Bay and the Sacramento River have been tagged with radio transmitters, and UC Davis researchers have placed over 30 monitoring stations along the Sacramento River to record their movements. However, adult green sturgeon only spawn every few years, so tagged fish may not return to spawn before the battery life of radio transmitters expires. Though these radio telemetry studies will eventually improve our understanding of green sturgeon migration patterns and general habitat use, they may produce only small datasets.

Researchers in the Rogue River have observed green sturgeon holding in deep pools both prior to and after spawning. These pools are generally located downstream of suspected spawning sites. Anecdotal evidence from recreational fishing suggests that green sturgeon may also exhibit similar pre- and post-spawning holding phases in the Sacramento River (J. Heublein, personal communication, 14 November, 2005). Expanding existing radio telemetry studies by capturing and tagging additional fish will increase the likelihood of tracking the movement of spawners in the Sacramento River. To increase the chance of tracking a spawning migration before the batteries in radio transmitters expire, future tagging efforts should focus on capturing adult green sturgeon during a period when, and in locations where, they are suspected of exhibiting pre-spawning holding behavior in the Sacramento River. It may also be possible to capture adults that congregate below RBDD once the gates are lowered on 15 May.

Current radio telemetry studies rely primarily on the remote monitoring stations that UC Davis researchers have placed along the Sacramento River. By tracking the upstream migration runs of spawning adults, these remote stations will yield valuable information on migration timing and rate. The spacing of the remote monitoring stations will also help to bracket potential holding and spawning sites. However, conservation and restoration of green sturgeon requires a better understanding of the specific habitats they use in the Sacramento River. Based on habitat preferences of green sturgeon that spawn in the Rogue and Klamath rivers (Erickson et al. 2002), we hypothesize that turbulent reaches in Iron Canyon, including China Rapids, may serve as important green sturgeon spawning sites. To better identify specific spawning habitats, we recommend that state and federal agencies supplement the existing radio telemetry studies, which rely on stationary recorders, with more strategic and active monitoring using hand-held equipment that focuses on suspected habitats like the turbulent reaches of Iron Canyon. The same approach could be used to identify potential pre- and post-spawning holding sites in the middle Sacramento River.

The protected status of the southern DPS of green sturgeon may compel NMFS to limit the number of green sturgeon that can be handled and tagged in the Sacramento River. If tagging of adults is curtailed in the Sacramento River system, we recommend that state and federal agencies commission studies in the Rogue and Klamath river systems designed specifically to identify spawning habitat preferences. This knowledge could be transferred to the Sacramento River corridor to assist the process of identifying likely spawning sites, which could then be monitored using methods generally approved for listed fish species (e.g., SCUBA or snorkeling).

10.3.7 Survey western pond turtle distribution and abundance

Little is known about the basic distribution and abundance of western pond turtle in the Sacramento River basin, or the habitats that they use in the river corridor. Based on research conducted in other river systems, we hypothesize that off-channel habitats are likely to be more important to this species in the Sacramento River corridor than mainstem habitats. Existing uncertainties about western pond turtle populations and their local habitat needs greatly limit our ability to development effective management strategies to conserve this species of special concern.

A one-time comprehensive or synoptic survey of western pond turtle distribution and abundance, combined with more focused surveys to identify habitat associations for each key life stage, would allow us to test a number of hypotheses about factors currently limiting western pond turtle populations and the relative benefits of implementing particular management actions.

The synoptic survey should include a systematic assessment of western pond turtle distribution and abundance along the Sacramento River from Keswick Dam to the Delta. Ideally this would include a comprehensive survey of all potential western pond turtle aquatic habitats in the river corridor. However, access to private property, cost, and other logistical constraints might require adopting a less intensive effort that would focus on surveying representative habitat units and sub-reaches. Subsequent to the survey, a subset of habitat units and sub-reaches could be selected as representative "index" sites for periodic resurveying as part of a longer term trends monitoring effort.

In addition to the baseline survey described above, the following types of focused studies on the western pond turtle in the Sacramento River corridor would also be beneficial:

- Habitat and microhabitat associations for rearing of hatchlings and juveniles.
- Female nest site selection behavior and mapping of nest site locations with associated habitat information (relative elevation, distance from nearest water body, soil texture, slope, aspect, vegetation, etc.).
- Year-round movement patterns of adults and juveniles to examine:
    - Local seasonal movement patterns in terrestrial and aquatic environments,
    - Broad-scale dispersal patterns in both terrestrial and aquatic environments,
    - Impacts of flood events with varying magnitudes, timings and durations.
- Phenology (timing) of nesting, incubation, hatching, and hatchling emergence and dispersal from the nest.

10.3.8 Conduct long-term monitoring of cottonwood recruitment and riparian vegetation dynamics

The recent studies of cottonwood seedling initiation and establishment conducted by TNC (2003) and DWR (Morgan and Henderson 2005a, 2005b) have calibrated the recruitment box model for application to three point bars in the Middle Sacramento River (RM 192, 183, 172). Additional focused studies on other point bar sites may be warranted to expand the geographic range of calibrated sites so that flow management actions could better be directed to promote cottonwood recruitment in any particular sub-reach or site from Red Bluff to Colusa. Combining and extending the work by Wood (2003a) with the relative elevation model developed by Greco and others (Greco et al. in press) might be a valuable and cost-effective means of achieving this goal.

A number of hypotheses and conceptual models concerning riparian plant community succession in the Central Valley have been proposed over the past several decades (see Chapter 9), yet we still have relatively little data with which to test these hypotheses. The results of several recent studies (Fremier 2003, Vaghti 2003, and Wood 2003b) suggest first, that riparian vegetation dynamics are likely much more complex and less deterministic than indicated by the classic succession model (see Section 9.3), and second, that the relative abundance and recruitment rates of certain dominant riparian tree species have been altered in recent decades, presumably due to hydrologic alterations and other human impacts on the river-riparian ecosystem. Building on the preliminary work of Wood (2003b) by continuing to monitor the 18 permanent vegetation plots he established, plus adding additional plots along the Middle Sacramento River, would allow us to improve our understanding of:

- Long-term patterns in recruitment and establishment of cottonwood and other native riparian trees and shrubs,
- Amount and rate of expansion of non-native invasive plant species, and
- Alterations in riparian vegetation dynamics and patterns of plant community succession.

Our current understanding of riparian vegetation conditions and the effects of various potential management actions is limited by the nature of the existing mapped vegetation data. Although the existing mapped information is useful, it was compiled over a number of years and is now in need of updating to reflect changes since 1997. In addition, recent advancements in remote sensing technology and vegetation classification approaches would allow us to develop a more refined, hierarchical classification approach (see Tu 2000 and Vaghti 2003 for examples) that could be combined with newer high resolution imagery (e.g., LIDAR and hyperspectral imagery) to produce an updated, fine-scale vegetation map for the river and create the foundation for efficient future updates at regular intervals (e.g., every 5 or 10 years). This improved vegetation mapping could then be used to track changes in habitat for various species, such as riparian songbirds, and be used in modeling potential effects of various potential flow and land management actions.

10.4 Acknowledgements

Funding for this SOS Report was provided by the CALFED Ecosystem Restoration Program and the Packard Foundation, both of which were administered by The Nature Conservancy. The authors would also like to thank the numerous experts that contributed source material, ideas, and feedback to the development of this report.

10.5 References

Bigelow, P. E. 1996. Evaluation of the Sacramento River spawning gravel restoration project and winter-run Chinook salmon redd survey, 1987-1993. Final report. U.S. Fish and Wildlife Service, Northern Central Valley Fish and Wildlife Office, Red Bluff, California.

Bjornn, T. C. 1971. Trout and salmon movements in two Idaho streams as related to temperature, food, stream flow, cover, and population density. Transactions of the American Fisheries Society 100: 423-438.

Buer, K. 1984. Middle Sacramento River spawning gravel study. California Department of Water Resources, Northern District, Red Bluff.

Buer, K. 1985. Sacramento River spawning gravel studies. Executive Summary, Record No. ES-22. California Department of Water Resources, Northern District, Red Bluff.

Buer, K. 1994a. Use of alternative gravel sources for fishery restoration and riparian habitat enhancement in Shasta and Tehama Counties, California. California Department of Water Resources, Northern District, Red Bluff.

Buer, K. 1994b. Sacramento River bank erosion investigation memorandum progress report. Internal memorandum to R. Scott and L. Brown from K. Buer, Chief, Geology Section, California Department of Water Resources, Northern District, Red Bluff.

Burner, C. J. 1951. Characteristics of spawning nests of Columbia River salmon. U.S. Fish and Wildlife Service Fishery Bulletin 52: 97-110.

Bustard, D. R., and D. W. Narver. 1975. Aspects of the winter ecology of juvenile coho salmon (*Oncorhynchus kisutch*) and steelhead trout (*Salmo gairdneri*). Journal of the Fisheries Research Board of Canada 32: 667-680.

California Resources Agency. 1989. Upper Sacramento River Fisheries and Riparian Habitat Management Plan. Prepared by SB 1086 Advisory Council.

CDFG (California Department of Fish and Game). 1983. Sacramento River and tributaries bank protection and erosion control investigation--evaluation of impacts on fisheries. Final Report. CDFG, Bay-Delta Fishery Project.

CDFG (California Department of Fish and Game). 1992. Recovery plan: bank swallow. Report No. 93.02. CDFG, Nongame Bird and Mammal Section, Wildlife Management Division, Sacramento.

CDFG (California Department of Fish and Game). 1997. Central Valley anadromous fish-habitat evaluations: Sacramento and American river investigations, October 1995 through September 1996. Stream Evaluation Program, Technical Report No. 97-1. Prepared by CDFG, Environmental Services Division, Stream Flow and Habitat Evaluation Program for U.S. Fish and Wildlife Service, Central Valley Anadromous Fish Restoration Program.

CDFG (California Department of Fish and Game). 1998. Central Valley anadromous fish-habitat evaluations: October 1996 through September 1997. Stream Evaluation Program, Technical Report No. 98-4. Prepared by CDFG, Environmental Services Divsion, Stream Flow and Habitat Evaluation Program for U.S. Fish and Wildlife Service, Central Valley Anadromous Fish Restoration Program.

CDFG (California Department of Fish and Game). 1999. Central Valley anadromous fish-habitat evaluations: October 1997 through September 1998. Stream Evaluation Program, Technical Report No. 99-4. Prepared by CDFG, Water and Aquatic Habitat Conservation Branch for U.S. Fish and Wildlife Service, Central Valley Anadromous Fish Restoration Program.

CDFG (California Department of Fish and Game). 2000. Central Valley anadromous fish-habitat evaluations: October 1998 through September 1999. Stream Evaluation Program, Technical Report No. 00-8. Prepared by CDFG, Habitat Conservation Division, Native Anadromous Fish and Watershed Branch for U.S. Fish and Wildlife Service, Central Valley Anadromous Fish Restoration Program.

CDFG (California Department of Fish and Game). 2002. Sacramento River winter-run Chinook salmon. Biennial Report 2000-2001. Prepared by CDFG, Habitat Conservation Division, Native Anadromous Fish and Watershed Branch for California Fish and Game Commission.

CDFG (California Department of Fish and Game). 2004. Sacramento River winter-run Chinook salmon. Biennial Report 2002-2003. Prepared by CDFG, Habitat Conservation Division, Native Anadromous Fish and Watershed Branch for California Fish and Game Commission.

Clark, G. H. 1929. Sacramento-San Joaquin salmon (*Oncorhynchus tshawytscha*) fishery of California. California Department of Fish and Game Fish Bulletin 17: 1-73

Cramer, S. P., and D. B. Demko. 1997. The status of late-fall and spring chinook salmon in the Sacramento River basin regarding the Endangered Species Act. Special Report. Submitted to National Marine Fisheries Service on behalf of Association of California Water Agencies and California Urban Water Agencies. Prepared by S. P. Cramer and Associates, Inc., Gresham, Oregon.

Cui, Y. (in press). The Unified Gravel-Sand (*TUGS*) Model: Simulating Sediment Transport and Gravel/Sand Grain Size Distributions in Gravel-Bedded Rivers, submitted to Water Resources Research.

DeHaven, R. W. 2000. Impacts of riprapping to ecosystem functioning, lower Sacramento River, California. Prepared by USFWS, Sacramento Office, Sacramento, California for U.S. Army Corps of Engineers, Sacramento District, Sacramento, California.

EA Engineering (EA Engineering, Science, and Technology). 1992. Lower Tuolumne River spawning gravel availability and superimposition. Appendix 6 to Don Pedro Project Fisheries Studies Report (FERC Article 39, Project No. 2299). *In* Report of Turlock Irrigation District and Modesto Irrigation District Pursuant to Article 39 of the License for the Don Pedro Project, No. 2299. Vol. IV. EA Engineering, Lafayette, California.

Erickson, D. L., J. A. North, J. E. Hightower, J. Weber, and L. Lauck. 2002. Movement and habitat use of green sturgeon *Acipenser medirostris* in the Rogue River, Oregon, USA. Journal of Applied Ichthyology 18: 565-569.

Everest, F. H., G. H. Reeves, J. R. Sedell, J. Wolfe, D. Hohler, and D. A. Heller. 1986. Abundance, behavior, and habitat utilization by coho salmon and steelhead trout in Fish Creek, Oregon, as influenced by habitat enhancement. Annual Report 1985 Project No. 84-11. Prepared by U.S. Forest Service for Bonneville Power Administration, Portland, Oregon.

Fisher, F. W. 1994. Past and present status of Central Valley chinook salmon. Conservation Biology 8: 870-873.

Fris, M. B., and R. W. DeHaven. 1993. A community-based habitat suitability index model for Shaded Riverine Aquatic Cover, selected reaches of the Sacramento River system. U.S. Fish and Wildlife Service, Sacramento Field Office, Sacramento, California.

Golet G. H., M. D. Roberts, E. W. Larsen, R. A. Luster, R. Unger, G. Werner, and G. G. White. 2006. Assessing societal impacts when planning restoration on large alluvial rivers: A case study of the Sacramento River Project, California. Environmental Management 37:862-879.

Garrison, B. A. 1999. Bank swallow (*Riparia riparia*). No. 414. A. Poole and F. Gill, editors. The birds of North America. The Academy of Natural Sciences, Philadelphia, Pennsylvania and The American Ornithologists' Union, Washington, D. C.

Hallock, R. J., and F. W. Fisher. 1985. Status of the winter-run chinook salmon, *Oncorhynchus tshawytscha*, in the Sacramento River. Anadromous Fisheries Branch Office Report. California Department of Fish and Game.

Hallock, R., and H. Rectenwald. 1990. Environmental factors contributing to the decline of the winter-run chinook salmon on the upper Sacramento River. Pages 141-145 in T. J. Hassler, editor. Proceedings of the 1990 Northeast Pacific chinook and coho salmon workshop. California Cooperative Fishery Research Unit, Humboldt State University, Arcata.

Hanson, H. A., O. R. Smith, and P. R. Needham. 1940. An investigation of fish-salvage problems in relation to Shasta Dam. Special Scientific Report No. 10, United States Department of the Interior, Bureau of Fisheries, Washington, DC.

Hartman, G. F. 1965. The role of behavior in the ecology and interaction of underyearling coho salmon (*Oncorhynchus kisutch*) and steelhead trout (*Salmo gairdneri*). Journal of the Fisheries Research Board of Canada 22: 1035-1081.

Healey, M. C. 1991. Life history of chinook salmon (*Oncorhynchus tshawytscha*). In C. Margolis and L. Groot, editors, *Pacific salmon life histories*, pp. 311–394. University of British Columbia Press, Vancouver.

Larsen, E. W., E. H. Girvetz, and A. K. Fremier. 2006. Assessing the effects of alternative setback channel constraint scenarios employing a river meander migration model. Environmental Management 37: 880-897. doi:10.1007/s00267-004-0220-9.

Limm, M. P., and M. P. Marchetti. 2003. Contrasting patterns of juvenile Chinook salmon (*Oncorhynchus tshawytscha*) growth, diet, and prey densities in off-channel and main stem habitats on the Sacramento River. Prepared for The Nature Conservancy, Chico, California.

Lindley, S. T., R. Schick, B. P. May, J. J. Anderson, S. Greene, C. Hanson, A. Low, D. McEwan, R. B. MacFarlane, C. Swanson, and J. G. Williams. 2004. Population structure of threatened and endangered Chinook salmon ESUs in California's Central Valley Basin. Technical Memorandum NOAA-TM-NMFS-SWFSC-360. National Marine Fisheries Service, Southwest Fisheries Science Center.

Lytle, D. A., and N. L. Poff. 2004. Adaptation to natural flow regimes. Trends in Ecology & Evolution 19: 94–100.

Maslin, P., M. Lennox, J. Kindopp, and W. McKinney. 1997. Intermittent streams as rearing habitat for Sacrametno River Chinook salmon *(Oncorhynchus tshawytscha)*. Department of Biological Sciences, California State University.

McBain and Trush. 2002. San Joaquin River restoration study background report. Prepared for Friant Water Users Authority, Lindsay, California and Natural Resources Defense Council, San Francisco, California by McBain and Trush, Arcata.

Micheli, E. R., and E. W. Larsen. In preparation. River channel cutoff dynamics, Sacramento River, California, USA.

Moffatt, K. C., E. E. Crone, K. D. Holl, R. W. Schlorff, and B. A. Garrison. 2005. Importance of hydrologic and landscape heterogeneity for restoring bank swallow (*Riparia riparia*) colonies along the Sacramento River, California. Restoration Ecology 13: 391-402

Morgan, T. L. 2005. Hydrological and physiological factors controlling Fremont cottonwood seedling establishment along the Sacramento River, California: surface and alluvial groundwater relations, ecophysiological analysis from 2002-2004 field seasons, and cottonwood water use determination using stable isotope analysis. Master's thesis. California State University, Chico.

Morgan, T. L., and A. Henderson. 2005a. Field observations of cottonwood seedling survival at River Mile 192.5 during 2002 and 2003, Sacramento River, California. 30 March. Memorandum report. Prepared for California Department of Water Resources, Northern District.

Morgan, T. L., and A. Henderson. 2005b. Cottonwood seedling monitoring during 2004 and 2005 along the Sacramento River, California. 30 December. Memorandum draft report. Prepared for California Department of Water Resources, Northern District.

Morken, I., and G. M. Kondolf. 2003. Evolution assessment and conservation strategies for Sacramento River oxbow habitats. Prepared for The Nature Conservancy, Sacramento River Project. Berkeley, California.

Moyle, P. B. 2002. Inland fishes of California. Revised edition. University of California Press, Berkeley.

NMFS (National Marine Fisheries Service). 1993. Biological opinion for the operation of the federal Central Valley Project and the California State Water Project. Prepared for U.S. Bureau of Reclamation, Sacramento, California by NMFS, Southwest Region, Long Beach, California.

NMFS. 1997. NMFS Proposed recovery plan for the Sacramento River winter-run chinook salmon. NMFS, Southwest Region, Long Beach, California.

NMFS. 2004. Biological opinion on the long-term Central Valley Project and State Water Project operations criteria and plan. Endangered Species Act Section 7 Consultation. NMFS, Southwest Region, Long Beach, California.

NMFS. 2006. Endangered and threatened wildlife and plants: threatened status for Southern Distinct Population Segment of North American green sturgeon: final rule. Federal Register 71: 17757-17766.

Roberts, M. D., D. R. Peterson, D. E. Jukkola, and V. L. Snowden. 2002. A pilot investigation of cottonwood recruitment on the Sacramento River. The Nature Conservancy, Sacramento River Project, Chico, California.

Rood, S. B., and J. M. Mahoney. 2000. Revised instream flow regualtion enables cottonwood recruitment along the St. Mary River, Alberta, Canada. Rivers 7: 109–125.

Rood, S. B., C. R. Gourley, E. M. Ammon, L. G. Heki, J. R. Klotz, M. L. Morrison, D. Mosley, G. G. Scoppettone, S. Swanson, and P. L. Wagner. 2003. Flows for floodplain forests: a successful riparian restoration. BioScience 53: 647-656.

Rood, S. B., G. M. Samuelson, J. H. Braatne, C. R. Gourley, F. M. R. Hughes, and J. M. Mahoney. 2005. Managing river flows to restore floodplain forests. Frontiers in Ecology and the Environment 3: 193-201.

SJRGA (San Joaquin River Group Authority). 2005. 2004 Annual technical report on the implementation and monitoring of the San Joaquin River Agreement and the Vernalis Adaptive Management Plan.

Slater, D. W. 1963. Winter-run chinook salmon in the Sacramento River, California with notes on water temperature requirements at spawning. Special Scientific Report—Fisheries 461. U.S. Fish and Wildlife Service.

Snider, B., B. Reavis, and S. Hill. 1998. Upper Sacramento River winter-run Chinook salmon escapement survey, April-August 1997. Stream Evaluation Program Technical Report No. 98-1. California Department of Fish and Game, Environmental Services Division.

Snider, B., B. Reavis, and S. Hill. 1999. Upper Sacramento River winter-run Chinook salmon escapement survey, May-August 1998. Stream Evaluation Program Technical Report No. 99-1. California Department of Fish and Game, Water and Aquatic Habitat Conservation Branch.

Snider, B., B. Reavis, and S. Hill. 2000a. Upper Sacramento River winter-run Chinook salmon escapement survey, May-August 1999. Stream Evaluation Program Technical Report No. 00-1. California Department of Fish and Game, Habitat Conservation Division, Native Anadromous Fish and Watershed Branch.

Snider, B., B. Reavis, and S. Hill. 2000b. Upper Sacramento River fall-run Chinook salmon escapement survey, September-December 1999. Stream Evaluation Program Technical Report No. 00-3. California Department of Fish and Game, Habitat Conservation Division, Native Anadromous Fish and Watershed Branch.

Snider, B., B. Reavis, and S. Hill. 2000c. Upper Sacramento River late-fall-run Chinook salmon escapement survey, December 1999-April 2000. Stream Evaluation Program Technical Report No. 00-9. California Department of Fish and Game, Habitat Conservation Division, Native Anadromous Fish and Watershed Branch.

Snider, B., B. Reavis, and S. Hill. 2001. Upper Sacramento River winter-run Chinook salmon escapement survey, May-August 2000. Stream Evaluation Program Technical Report No. 01-1. California Department of Fish and Game, Habitat Conservation Division, Native Anadromous Fish and Watershed Branch.

Sommer, T. R., M. L. Nobriga, W. C. Harrell, W. Batham, and W. J. Kimmerer. 2001. Floodplain rearing of juvenile chinook salmon: evidence of enhanced growth and survival. Canadian Journal of Fisheries and Aquatic Sciences 58: 325-333.

SRCAF (Sacramento River Conservation Area Forum). 2003. Sacramento River Conservation Area Forum handbook. Prepared for The Resources Agency, State of California by the Sacramento River Advisory Council; revised and updated by the Sacramento River Conservation Area Forum, Red Bluff, California. http://www.sacramentoriver.ca.gov.

Stillwater Sciences, W. Rainey, E. Pierson, C. Corben, and M. Power. 2003. Sacramento River ecological indicators pilot study. Prepared by Stillwater Sciences, Berkeley, California for The Nature Conservancy, Chico, California.

Stillwater Sciences. 2006. Restoring recruitment processes for riparian cottonwoods and willows: a field-calibrated predictive model for the lower San Joaquin Basin. Prepared for CALFED Bay-Delta Ecosystem Restoration Program, Sacramento, California by J. Stella, Stillwater Sciences, Berkeley, California, in conjunction with J. Battles and J. McBride, Department of Environmental Science, Policy, and Management, University of California, Berkeley.

Swales, S., R. B. Lauzier, and C. D. Levings. 1986. Winter habitat preferences of juvenile salmonids in two interior rivers in British Columbia. Canadian Journal of Zoology 64: 1506-1514.

TNC (The Nature Conservancy). 2003. Beehive Bend Addendum to: A pilot investigation of cottonwood recruitment on the Sacramento River. The Nature Conservancy, Sacramento River Project, Chico, California.

TNC (The Nature Conservancy). 2005. Restoring biodiversity health on the middle Sacramento River. The Nature Conservancy, Chico, California.

Tu, I.-Y. M. 2000. Vegetation and processes of natural regeneration in periodically flooded riparian forests in the Central Valley of California. Doctoral dissertation. University of California, Davis.

USBR (U.S. Bureau of Reclamation). 2004. Long-term Central Valley Project and State Water Project Operations Criteria and Plan Biological Assessment. USDI Bureau of Reclamation, Mid-Pacific Region, Sacramento, California.

USFWS (U.S. Fish and Wildlife Service). 1995. Working paper on restoration needs: habitat restoration actions to double natural production of anadromous fish in the Central Valley of California. Volume 3. May 9, 1995. Prepared for USFWS under the direction of the Anadromous Fish Restoration Program Core Group, Stockton, California.

USFWS (U.S. Fish and Wildlife Service). 2005a. Flow-habitat relationships for macroinvertebrates in the Sacramento River between Keswick Dam and Battle Creek. Draft report. USFWS, Sacramento Fish and Wildlife Office, Sacramento, California.

USFWS (U.S. Fish and Wildlife Service). 2005b. Flow-habitat relationships for Chinook salmon rearing in the Sacramento River between Keswick Dam and Battle Creek. USFWS, Sacramento Fish and Wildlife Office, Sacramento, California.

Vaghti, M. G. 2003. Riparian vegetation classification in relation to environmental gradients, Sacramento River, California. Master's Thesis, University of California, Davis

Vogel, D., and G. Taylor. 1987. Survey of the chinook salmon spawning substrate in the Sacramento River from the Highway 273 bridge to Keswick Dam, July - August, 1987. U.S. Fish and Wildlife Service, Fishery Assistance Office, Red Bluff, California.

Vogel, D. A., and K. R. Marine. 1991. Guide to upper Sacramento River chinook salmon life history. Prepared for the U.S. Bureau of Reclamation, Central Valley Project.

Vogel, D. A., K. R. Marine, and J. G. Smith. 1988b. Fish passage action program for Red Bluff Diversion Dam. Final report on fishery investigations. FR1/FAO-88-19. U.S. Fish and Wildlife Service, Fisheries Assistance Office, Red Bluff, California.

Vronskiy, B. B. 1972. Reproductive biology of the Kamchatka River chinook salmon (*Oncorhynchus tshawytscha*). Walbaum J. Icthyol. 12:259-273.

Yoshiyama, R. M., F. W. Fisher, and P. B. Moyle. 1998. Historical abundance and decline of Chinook salmon in the Central Valley region of California. North American Journal of Fisheries Management 18: 487–521.

Yoshiyama, R. M., E. R. Gerstung, F. W. Fisher, and P. B. Moyle. 1996. Historical and present distribution of Chinook salmon in the Central Valley drainage of California. Sierra Nevada Ecosystem Project, Final Report to Congress, Volume 3. University of California.

Yoshiyama, R. M., E. R. Gerstung, F. W. Fisher, and P. B. Moyle. 2001. Historical and present distribution of chinook salmon in the Central Valley drainage of California. Pages 71-176 in R. L. Brown, editor. Contributions to the biology of Central Valley salmonids. Fish Bulletin 179: Volume 1. California Department of Fish and Game, Sacramento.