RECLAMATION
Managing Water in the West

Piscine Predators and Control Measures: Application to In-river Structures in the San Joaquin River Restoration Area

U.S. Department of the Interior
Bureau of Reclamation
Technical Service Center
Denver, Colorado

January 2014

MISSION STATEMENTS

The U.S. Department of the Interior protects America's natural resources and heritage, honors our cultures and tribal communities, and supplies the energy to power our future.

The mission of the Bureau of Reclamation is to manage, develop, and protect water and related resources in an environmentally and economically sound manner in the interest of the American public.

Piscine Predators and Control Measures: Application to In-river Structures in the San Joaquin River Restoration Area

Prepared by

Jarod D. Hutcherson
Water and Environmental Resources Division
Fisheries and Wildlife Resources Group

U.S. Department of the Interior
Bureau of Reclamation
Technical Service Center
Denver, Colorado

January 2014

CONTENTS

Figures

ABSTRACT

A primary objective of restoration efforts on the San Joaquin River is to restore and maintain fish populations from Friant Dam to the confluence of the Merced River. Native and nonnative piscivorous fish are currently present in the wetted sections of the upper San Joaquin River. The San Joaquin River Restoration Program recognizes restoration efforts will potentially add structures within the river that could provide additional areas for predator holding, which may hinder recovery effectiveness. This paper reviews available information, including direct and indirect control methods, to evaluate potential techniques to reduce predation at in-river structures and facilitate restoration efforts. Direct control methods include: direct removal of target fish by angling, netting, or electrical removal, eradication by chemicals, and isolating problematic fish from native fish populations. Unfortunately, information regarding the success of these programs is far from conclusive. Most previous studies focus on immediate results and fail to acknowledge whether programs are sustainable, or can remove sufficient numbers of predators to increase native fish recruitment. However, this does not preclude the use of these predator control methods. During initial efforts, direct control may facilitate reintroduction of salmon and other native fishes in the Restoration Area by reducing immediate predation pressure. Long-term efforts should be directed at increasing habitat suitable for native fish while concurrently reducing habitat that favors nonnative species. While direct control efforts are not likely sustainable, they should be included as part of broader restoration efforts, facility improvements, and regulatory changes.

INTRODUCTION

The San Joaquin River Restoration Program (SJRRP) was established following the settlement between the Natural Resources Defense Council, *et al.* v. Kirk Rodgers, *et al.* (NRDC 2006). The SJRRP has organized efforts to restore flows to the San Joaquin River while minimizing adverse impacts to water users in the area. Furthermore, an agreement in the settlement stipulates that the SJRRP “maintain fish populations in ‘good condition’ in the main stem of the San Joaquin River below Friant Dam to the confluence of the Merced River.” This includes reintroduction efforts to restore Chinook salmon (*Oncorhynchus tshawytscha*) that were previously extirpated from this section of the river, following construction of Friant Dam in 1942. While native and nonnative predators are already present in the Restoration Area (Figure 1), the SJRRP recognizes construction/restoration projects may add structures to the San Joaquin River that may provide additional habitat where predators could reside. In addition, current structures (*e.g.*, dams, diversions, mine pits) on the San Joaquin may harbor predators that will be detrimental to restoration efforts. Reducing or controlling piscivorous fishes at in-river structures will reduce predation pressure on juvenile salmon and other native fishes. This paper is a synthesis of available research to determine what methods could potentially be applied in the Restoration Area, in order to reduce piscivorous predation, at and near in-river structures.

PISCIVOROUS FISH OF THE SAN JOAQUIN RIVER

Over the last century, native fish distribution in the San Joaquin River has changed dramatically; particularly after construction of Friant Dam in 1942, river conditions now support nonnative species and native species distribution has diminished (Moyle 1973). Brown sampled fish communities in the lower San Joaquin River, including parts of the Merced, Tuolumne, and Stanislaus river from 1993-1995 (2000). He noted that native fish can persist below major foothill dams but nonnative fish predominate in lower stretches of the river, and suggested that habitat in these lower reaches of the river may provide unsuitable spawning areas for native fishes. Sampling efforts by the SJRRP indicated increasing temperatures, turbidity, and conductivity, from upstream to downstream reaches within the Restoration Area, coincident with the abundance of nonnative species (unpublished data).

Piscivorous fish in the Restoration Area include rainbow trout (*O. mykiss*), brook trout (*Salvelinus fontinalis*) Sacramento pikeminnow (*Ptychocheilus grandis*), striped bass (*Morone saxatilis*), black bass (*Micropterus* spp.), other centrarchids

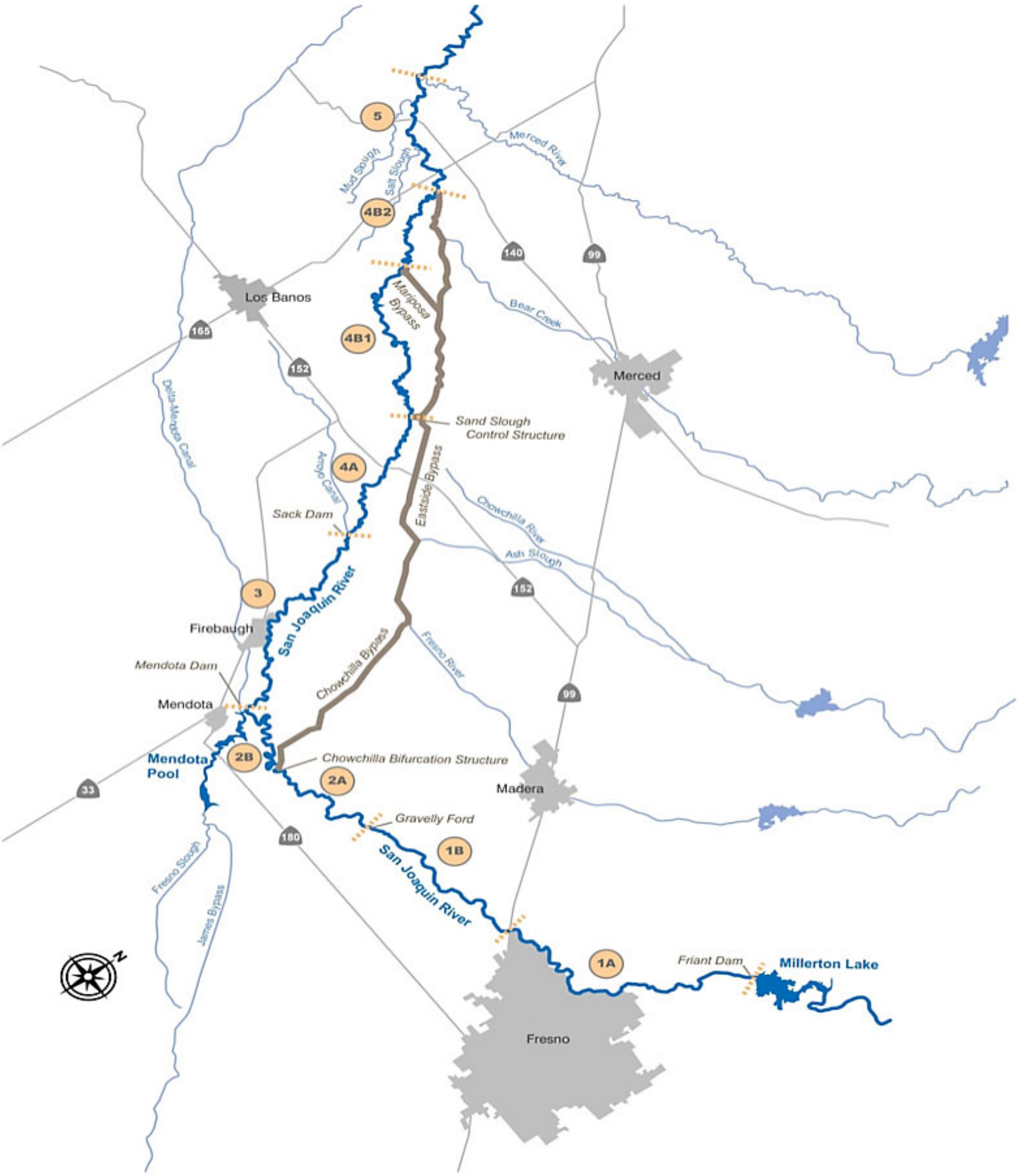

Figure 1.—Map of the San Joaquin River Restoration Area and associated reaches. The yellow circles identify the associated restoration reach, and the dashed-yellow line identifies the separation between the labeled reaches. The area encompasses the San Joaquin River below Friant Dam to the confluence of the Merced River.

including sunfish (*Lepomis* spp.) and crappie (*Proxomis* spp.), catfish (*Ameiurus* spp. and *Ictalurus punctatus*), and sculpin (*Cottus* spp.; SJRRP 2013, *in draft*). Of these fish, only pikeminnow, rainbow trout, and sculpin are native (Moyle 2002; Brown and Michniuk 2007).

Native Piscivores

Sacramento pikeminnow is a large, piscivorous cyprinid of the Sacramento-San Joaquin River system, most abundant in lightly disturbed areas of low velocity, deep pools, and overhanging vegetation (Moyle 2002). Invertebrates generally make up the majority of diet for pikeminnow < 150 mm, but become piscivorous with increasing length (Nobriga *et al.* 2006). Pikeminnow typically may not consume salmonids in great numbers (Nobriga *et al.* 2006; Nobriga and Feyrer 2007); however, pikeminnow have been observed holding below structures in the Sacramento-San Joaquin River system and take advantage of seasonally abundant resources (*i.e.*, juvenile salmon; Kano 1987; Vogel *et al.* 1988). The reputation of the pikeminnow as the primary predator of salmon often stems from studies centered on the Columbia River system in the northwest U.S., where northern pikeminnow (*P. oregonensis*) contribute to the greatest proportion of juvenile salmon predation (Rieman *et al.* 1991; Vigg *et al.* 1991).

Rainbow trout in the Restoration Area are likely hatchery-stocked fish (California Department of Fish and Game; https://nrm.dfg.ca.gov/fishplants/); steelhead are largely absent in the Restoration Area (Portz *et al.* 2012). Rainbow trout are typically found in cool water with abundant riffles, while larger fish may reside in deep pools (Moyle 2002). Small rainbow trout (< 150 mm FL) primarily consume invertebrates while fish > 250 mm FL are primarily piscivorous (Beauchamp 1990). In other systems, salmonids may be primary predator of juvenile salmonids (Beauchamp 1995; Tabor *et al.* 2004b; Krueger *et al.* 2011). Like pikeminnow, large rainbow trout have been observed holding near in-river structures, preying on incoming salmonids (Odenweller, *pers. comm.*). However, little information is available that quantifies rainbow trout predation on juvenile Chinook salmon in California waterways.

Whether or not sculpin significantly contribute to predation on juvenile salmonids is debatable (Merz 2002; Tabor *et al.* 2004a, b). Sculpin were found to prey on sockeye salmon (*O. nerka*) fry migrating through lighted sections of the Cedar River in Washington (Tabor *et al.* 2004a). Likewise, sculpin consumed salmonid fry in laboratory experiments. In Lake Washington, Beauchamp (1995) notes that sculpins were rarely recovered during electrofishing efforts but were, nonetheless, “a significant component of the stream assemblage” and “should not be disregarded as potential predators.” Diets of prickly sculpin (*C. asper*) and Chinook salmon from the Mokelumne River were compared in 1998-1999 (Merz 2002). Sculpin diet consisted largely of invertebrates and larval fish, including Sacramento sucker (*Catostomus occidentalis*), cyprinids, centrarchids, and Pacific lamprey (*Lampetra tridentate*). However, Chinook salmon were not present in any of the fish sampled. Conversely, prickly sculpin *were* observed in the diets of juvenile Chinook salmon, along with other larval fish and invertebrates.

Nonnative Piscivores

Other piscivores in the San Joaquin River are composed of nonnative fishes. These fish, including brook trout, striped bass, catfish, black bass, and other centrarchids, are found throughout the Restoration Area (SJRRP 2013, *in draft*). While native fish are most abundant in Reach 1, nonnative fish make up the majority of fish sampled in all downstream reaches. In the agriculturally dominated areas of the Sacramento-San Joaquin Delta, Feyrer and Healey (2003) found that nonnative fish were frequently associated with low flow and warm water temperatures, while native fish abundance was positively correlated with flow; however, striped bass and white catfish (*A. catus*) are also positively associated with higher flows, though white catfish showed similar correlation to other nonnative fish with increasing water temperatures.

Brook trout are native to the eastern United States but are now widespread in distribution (Snyder 1940). Brook trout were introduced early in California in lakes and streams for their rapid growth and angler catchability (Reimers, 1955; Snyder 1940). While brook trout are opportunistic feeders, larger fish often prey on fish (Brown and Rasmussen 2009; East and Magnan 1991). When introduced, brook trout often mature rapidly and often outcompete native salmonids (Novinger and Rahel 2003; Dunham *et al.* 2002; Gresswell 1991).

Striped bass are a pelagic predator, originally native to the east coast that are generally opportunistic predators (Moyle 2002). Though opportunistic predators, striped bass are often considered primary piscine predators at facilities in the Sacramento-San Joaquin Delta (Liston *et al.* 1994; Gingras and McGee 1997; Nobriga and Feyrer 2007). Like pikeminnow, striped bass often hold below in-river structures and prey on incoming fish (Liston *et al.* 1994; Tucker *et al.* 1998).

Black bass distribution in the Central Valley is variable; largemouth bass (*M. salmoides*) generally tend to prefer warm, slow-moving waters with abundant vegetation; smallmouth bass (*M. dolomieu*) tend to prefer cooler, clearer waters with rocky bottoms; and spotted bass (*M. punctulatus*) have a distribution between the two, preferring faster waters than largemouth bass but are frequently found in turbid waters lacking aquatic vegetation (Moyle 2002). Smallmouth bass are much less abundant in the Restoration Area than either spotted or largemouth bass (SJRRP 2013, *in draft*).

Because of the reputation of black bass for consuming salmonids in the northwestern US, they are often labeled as significant predators of juvenile salmon in Central Valley streams and rivers, particularly at in-river mining pits (Reyenolds *et al.* 1993; Kondolf *et al.* 1996). In the northwest US, smallmouth and largemouth bass diet was evaluated for consumption of salmonids (Tabor *et al.* 2007). Stomach contents of captured fish were recovered by gastric lavage. Smallmouth bass consumed significantly more salmonids than largemouth bass, both in numbers and as total percentage of diet. Once large enough to become piscivorous, consumption of salmonids is inversely related to size, though consumption of fish, in general, is positively correlated (Poe *et al.* 1991; Vigg *et al.* 1991; Fritts and Pearsons 2004). While the majority of published data regarding black bass predation focuses on waterways in the northwest, primarily involving smallmouth bass, little information is available regarding black bass predation in California rivers (Poe *et al.* 1991; Rieman *et al.* 1991; Fritts and Pearsons 2004).

Though data suggest that catfish are opportunistic feeders, catfish can be active salmonid predators when abundant numbers of juvenile salmon are present (Poe *et al.* 1991; Vigg *et al.* 1991; Liston *et al.* 1994). For example, of diet contents of 33 white catfish examined at the Tracy Fish Collection Facility in California, only 1 of 33 catfish stomach contents included fish (Liston *et al.* 1994). Conversely, salmonids made up 60% of the stomach contents of channel catfish, sampled in the tailrace of McNary Dam in the Columbia River system (Poe *et al.* 1991). Like pikeminnow, piscivory in catfish is generally positively correlated with size (Poe *et al.* 1991). In John Day Reservoir, salmonids were a larger proportion of the diet of channel catfish below the dam while non-salmonids made up a larger portion of the diet of catfish in the remainder of the reservoir, suggesting catfish may take advantage of a seasonally-abundant, spatially-available food source (*i.e.*, juvenile salmon passing through dam structures; Vigg *et al.* 1991).

While relatively little data exists regarding predation upon larval fish by centrarchids, they may be potentially significant predators of early-life stage salmonids and other native fishes of the San Joaquin River. Though salmonid predation by green sunfish (*L. cyanellus*) has not been studied, green sunfish (76-84 mm TL) readily preyed upon Gila chub (*Gila intermedia*, up to 25 mm TL) in an Arizona study (Dudley and Matter 2000). Bluegill (*L. macrochirus*; 103 mm TL) and white crappie (*P. annularis*; 171 mm TL) have been shown to prey on larval fish (centrarchid *spp.*) in controlled settings (Kim and DeVries 2001). Adult bluegill appeared to be size selective towards larvae in the 8–12 mm range. White crappie, though, were not statistically discriminative when preying on available larval fish. However, Chinook salmon are typically ≥ 20 mm at hatching (Wang 2010). While it might be assumed that bluegill, in the size class tested in Kim and DeVries (2001), may not target Chinook salmon larvae as prey items, white crappie and green sunfish could potentially consume larval Chinook salmon.

DIRECT CONTROL

The San Joaquin River Restoration Area presents unique issues regarding re-introduction efforts of Chinook salmon and concurrent predator control efforts. Because salmon have been "largely extirpated" from the San Joaquin River since construction of Friant Dam in 1942, little data is available regarding salmon life history and interactions with current fish species in this waterway (NRDC 2006).

Predation at in-river structures is often higher than areas without structures (Hall 1979; Vogel *et al.* 1988; Rieman *et al.* 1991). Restoration efforts will potentially add structures to the San Joaquin River where predatory fish may reside, which could hinder the effectiveness of recovery efforts. Drawing on previous research, agencies of the SJRRP hope to employ predator control methods that will reduce predation at in-river structures and facilitate recovery efforts.

Angling

Angling by agency personnel is likely not an effective method to remove sufficient numbers of fish; other angling efforts are available to have greater potential to reduce target fish (Liston *et al.* 1994; Porter 2011). At the Tracy Fish Collection Facility (TFCF), the catch-per-unit-effort of angling was less than other available methods (such as netting fish from the secondary channel; Liston *et al.* 1994). However, though angling may be less effective than other control methods, it could be utilized in areas where public access would be restricted; for instance, angling in the public-restricted areas in the Columbia River system was utilized to remove predatory pikeminnow holding near dams (Porter 2011). Dams on the Columbia River, though, are much larger than structures in the Restoration Area; where certain methods could not be utilized on the Columbia River system (*e.g.*, dangerous flows near dam face and depths that could prevent netting or electrofishing), these methods would likely present more effective control options in the Restoration Area. On the other hand, public participation in angling efforts may be more effective than angling by agency personnel and this is a viable option for a fish control program (Zimmerman *et al.* 2000; Paul *et al.* 2003). These efforts are described in more detail under “Indirect Control: Regulatory changes/Public recruitment.”

Netting

Netting fish is often used in fisheries research for surveys (Marchetti and Moyle 2001) and fish removal (Wu and Bridges, *in draft*). Netting is frequently combined with other efforts, such as electroshocking, to contain fish in an area targeted for evaluation. Nets are generally cheaper than other equipment (*e.g.*, electrofishing or the cost of chemical treatment) and more portable (Knapp and Matthews 1998). Nets are easy to place directly in areas where fish can be targeted, though depth and vegetation/debris may make placement more difficult.

To reduce predation pressure on native mountain yellow-legged frog (*Rana mucosa*), gill nets were used in five small, mountain lakes in the Sierra Nevada range (Vredenburg 2004). Nonnative salmonids were targeted for eradication from these lakes. Over eight years of efforts, trout were successfully removed from three of the five lakes. Key characteristics of these lakes that allowed for successful removal of fish were that no populations existed upstream of the lake and migration by downstream populations was prevented by barriers (no deep pools below waterfalls that would allow trout to jump these barriers).

In a similar study, gill netting was selected as an alternative to rotenone treatment, to remove trout from Maul Lake in California (Knapp and Matthews 1998). Netting was selected because, unlike chemical treatment, there were no adverse effects to non-target organisms; trout were the only fish in the lake and complete removal of all fish was the primary goal. Maul Lake is a subalpine lake in the Sierra Nevada Mountain Range where trout were historically absent. Brook and rainbow trout were the target removal species. Like the study above (Vredenburg 2004), Maul Lake was effectively isolated from other fish populations downstream by barriers and no populations above Maul Lake were available for reintroduction. Gill nets were found effective on larger fish (>110 mm) but were ineffective at removing smaller size classes because these fish were able to swim through the net panels. Based on a set of criteria for trout removal (small water body with barriers to up- and downstream movement, and a low density of fish), only 15-20% of high mountain lakes in the Sierra Nevada Range would be eligible for these removal efforts. In an open system, eradication through netting would likely be impossible.

Netting near structures in the Restoration Area could remove piscivorous fish. Though netting generally targets fish of a certain size (*i.e.*, fish that cannot swim through the netting), it does not necessarily target certain species. The advantages of netting are that personnel are able to remove fish from the net, selecting which species to return to the river and which ones to remove. Netting is generally labor intensive and frequent netting would probably be required near in-river structures because re-colonization would likely be high from open sections of river. Fouling of nets can frequently be an issue in areas with high amounts of aquatic vegetation and debris. Frequent cleaning may be required for nets to function efficiently.

Isolation

Measures to isolate communities of fish to prevent adverse effects to native species have frequently been attempted, usually as one aspect of a broader strategy. Typically, isolation measures are used in small mountain streams to isolate populations of native salmonids from nonnative ones (Paul *et al.* 2003; Novinger and Rahel 2003). Sometimes, isolation measures have been attempted in lakes or coves, on a larger geographical scale, in order to try and propagate native endangered species (Mueller and Burke 2005). Results of these studies are variable, though, and isolation itself may present a different subset of issues. Novinger and Rahel (2003) evaluated populations of cutthroat trout (*O. clarki*) in Wyoming steams to determine differences in isolated populations from non-isolated control sections. Sections of stream were isolated using artificial barriers, and nonnative brook trout were removed by electroshocking. Following removal of nonnative trout, hatchery-reared cutthroat trout were also stocked in these sections. Streams were monitored four to seven years following these efforts. While removal efforts greatly reduced numbers of brook trout, they were not eliminated. Isolation of these stream sections and removal of nonnative fish did not correspond in increased recruitment of cutthroat trout. Conversely, it was found that "fewer cutthroat trout persisted in isolation that coexisted with brook trout." One hypothesis for these results was that isolated stream sections lacked critical resources needed for cutthroat trout recruitment. Another was that isolated sections put native fish at risk from inadequate levels of immigration, which could lead to loss of genetic diversity.

Razorback suckers (*Xyrauchen texanus*) were raised on Lake Mohave, in ephemeral and permanent ponds created by wave-induced beach erosion (Mueller and Burke 2005). In particular, Davis Cove (1.3 ha, 7-m deep) was isolated with an 8-mm mesh net from Lake Mohave and predator removal efforts commenced June 1992. In an effort to increase juvenile sucker survival, predators were removed by electrofishing, angling, and netting (gill, fyke, trammel, and seine). Following predator removal efforts, 10,000 juvenile suckers (avg. 68 mm) and 167, 17-30 cm suckers were stocked. Net breaches were discovered in 1993, and in 1995 the cove was isolated with an earthen berm and rotenone used to remove resident fish. Even with predator removal efforts, sucker survival continued to decline. In another isolated pond, Yuma Cove, on Lake Mohave, it was noted that crayfish and odonate larvae quickly increased in numbers following predatory fish removal. Furthermore, Mueller and Burke note that predator control efforts through electrofishing and netting are selective towards larger fish which can lead to a shift in greater numbers of smaller predators. In Davis Cove, a resultant increase in sunfish biomass (>285%) occurred after a 50% reduction of largemouth bass.

The concept of isolation was introduced here because it if often one of the primary aspects of successful fish control studies (and often a necessary one; Knapp and Matthews 1998; Novinger and Rahel 2003; Vredenburg 2004). However, isolation, as an attempt to control nonnative predators, or preserve native fish, is not likely a viable option in the Restoration Area, particularly at or near structures. The entire tract of river, from the Sacramento-San Joaquin Delta to Friant Dam must remain passable to Chinook salmon. Even if predatory fish were removed from the Restoration Area and it were possible to construct some barrier, at the confluence of the Merced River for instance, that would exclude nonnative fish downstream, while allowing up- and downstream passage of Chinook Salmon, nonnative fish recruitment would be available from other tributaries of the San Joaquin River (*e.g.*, Millerton Lake, Mud, Salt, and Fresno Slough, and return water from the Delta-Mendota Canal). Even with isolation techniques, reintroduction of nuisance species (accidental or otherwise) is not unheard of, and can hinder eradication/removal efforts (Lintermans and Raadik 2001).

Electrical Control

Electricity is frequently utilized for fish control; methods include backpack electroshocking (Larson *et al.* 1986), boat electroshocking (Cavallo *et al.* 2012), electric seines (Bestgen *et al.* 2007), electrical crowders (Svoboda and Horn 2013), or arrays (Kano 1987). While electroshocking is often used to survey fish communities, controlling populations by electrical control is more limited (Paul *et al.* 2003; Cavallo *et al.* 2012; SJRRP 2013, *in draft*). Electroshocking for eradication/removal of nonnative species is typically limited to small mountain streams (Shepard *et al.* 2002; Peterson *et al.* 2008). Electrofishing in these areas is typically combined with other control methods such as isolation to prevent repopulation of nuisance fish species.

From 1993 to 2000, nonnative brook trout, threating a native cutthroat trout population, were removed from White’s Creek in the Missouri River Basin, Montana (Shepard *et al.* 2002). Wooden barriers were constructed in-stream, below waterfalls, in the treated section of stream. These were designed to prevent the formation of scour holes below the waterfall that could provide a jumping pool for downstream trout populations. After the construction of these barriers, multiple-pass electroshocking was conducted to remove brook trout. Fish >100 mm were more effectively removed than smaller fish during these efforts. After removal efforts, cutthroat trout in the treated sections increased significantly. However, application of methods in this study may have limited applicability to the San Joaquin River Restoration Area. The section of White’s Creek targeted for removal had a wetted width averaging 2 m, and an average depth of 10 cm, much smaller than the San Joaquin River. Furthermore, this stream section was essentially a closed system to fish because wood barriers were constructed below the treated section to prevent upstream movement by brook trout.

Smallmouth bass and northern pike were removed using electric seines in the Yampa River, Colorado (Bestgen *et al.* 2007). After four years of efforts (2003–2006), though, a positive response by small-bodied native fish was not detected. However, populations of some small-bodied nonnative fish have increased, possibly because of a release in predation pressure from removed predators. Several theories have been provided for the failed recruitment of native fish: a sufficient number of predators were not removed to increase native fish recruitment; the current native fish population is insufficient for increases to be realized; a compensatory response by younger smallmouth bass has increased predation on smaller native fish; environmental factors may be affecting native fish populations.

In May 2010, Cavallo *et al.* (2012) conducted a predator control effort on the North Fork Mokelumne River. Predatory fish were removed by multiple-pass depletion electroshocking. A total of 641 predatory fish were removed; sunfish and black bass were the dominant predators removed (n=615). It was estimated that 91% and 83% of the predators susceptible to electrofishing were captured on two separate days. Acoustically-tagged salmon smolts (mean 102 mm) were released before and after predator removal in the test reach and a 2.0 km control reach. The aim of these efforts was to determine differences in survival between groups of released salmon. Survival increased from <80% to >99% after the first predator removal effort. However, survival decreased to pre-removal efforts after the second predator removal. Observations were two-fold; 1) intense, site-specific predator removal efforts can improve survival of juvenile fishes immediately following these efforts; 2) predator removal efforts may be short-lived. After only one week, reductions in predation were no longer realized, possibly because of an influx of new predators.

In 1993, in Tanner Creek near the Bonneville Hatchery, the effectiveness of pikeminnow removal by electroshocking was evaluated (Ledgerwood *et al.* 1994). Prior to, and following pikeminnow removals, groups of 100,000 juvenile Chinook salmon (coded wire-tagged) were released in Tanner Creek and the main stem Columbia River, near the Bonneville Hatchery. After the initial release of salmon, electrofishing efforts followed in Tanner Creek, and downstream ~ 6 km along the shores of the Columbia River. Over 27 h of electrofishing effort, 2,866 northern pikeminnow were removed. Salmon were recovered downstream before and after these removals. Differences between salmon released in Tanner Creek and those released in the Columbia main stem were compared to try to determine whether predator removal efforts resulted in an increase in downstream migrating salmon. While recovery of salmon from the Columbia main stem was higher than the group released in Tanner Creek before removal, this difference was diminished following electrofishing efforts. The disparity between recovery rates was thought to diminish as a possible result of predator removal efforts that targeted predators from the migration route of fish released in Tanner Creek. Ledgerwood *et al.* notes that "there was little indication that northern [pikeminnow] recolonized the Tanner Creek or adjacent transect areas immediately after release of juvenile salmon from Bonneville Hatchery."

The Hallwood-Cordua Fish Screen is located on the Yuba River in north-central California. The fish screen consists of a diversion channel which guides fish from the main channel (Kano 1987). An upstream trash rack prevents large debris from overloading a fish screen downstream. Near the fish screen, predation was higher than other areas of the channel; pikeminnow were cited as the primary predator contributing to salmonid losses (Hall 1979). Large schools of Sacramento pikeminnow have been observed holding in the facility. Early attempts to remove predators through angling, gill-netting, and electroshocking were unsuccessful. In 1981, a series of electrodes, installed in the diversion channel, designed to reduce predator holding, were evaluated (Kano 1987). A series of 48 vertically-hanging electrodes (aluminum conduit wired in series) were installed between the upstream trash rack and downstream fish screen. The system was designed to drive predators towards the bypass structure, located in the fish screen, where they could be removed from the system. Groups of marked-Chinook salmon were released before and after trials. Recovery rates of salmon before and after predator removal were compared to determine loss rates. A gill-net was stretched across the upstream-side of the trash rack before using the electrical system to prevent upstream escape of predators in the diversion channel. Electrofishing ensued until no predators were recovered in the bypass. Following predator removal efforts, salmon recovery improved by ~4%. However, Kano notes effects of predator removals being short-lived as pikeminnow, upstream of the diversion, immediately repopulated the bypass channel.

In Reclamation’s Hydraulics Laboratory (Denver, CO), an electrical fish crowder was evaluated for potential application at the Tracy Fish Collection Facility (TFCF) to assist in predator removal (Svoboda and Horn 2013). The crowder was composed of a Smith-Root Electrofisher (Model LR-24), attached to an electrical sequencer, and seven sets of electrodes; the electrodes were attached vertically inside of an acrylic flume. The electrical sequencer was designed at the Denver Technical Service Center, and pulsed-DC current successively to each set of electrodes. Electrical current from the crowder, as well as water current in the flume, was evaluated at varying levels to determine whether large fish (striped bass 285-590 mm FL) could be driven through the electrical field by avoidance, rather than taxis or tetanus. Concurrently, the electrical field and varying flows were evaluated to determine if current sufficient to drive large fish would have minimal impacts on smaller fish (juvenile Chinook salmon and rainbow trout, 88–108 mm FL). Current was supplied successively to each set of electrodes in a manner that was slower than water velocity inside the test flume; this would allow fish potentially stunned by the electrical current to float downstream instead of remaining in the electrical field. After exposure to the electrical crowder, survival was monitored.

While striped bass exhibited avoidance response to pulse widths tested (1.2 ms and 10 ms), avoidance behavior was greater at a pulse width of 10 ms. Water velocities evaluated (0.46 and 0.76 m/s) did not significantly change the behavior of the striped bass for the electrical currents tested. Over 72 h, 3 of 6 striped bass tested at 0.76 m/s, 10-ms pulse width did not survive. However, mortality was attributed to handling stress over multiple experiments. Juvenile rainbow trout and Chinook salmon exhibited slight twitch behavior during testing. However, fish did not exhibit crowding behavior and typically remained within the electrical field. No mortality in juvenile trout or salmon occurred (N=60) over the following 72-h observation period.

Like mechanical control methods, electroshocking has its advantages: its effects are typically short-term and limited to the area immediately surrounding the electrical field. However, its drawbacks are that it has limited use in deeper water, or in low conductivity water, not all species are easily targeted by electroshocking (benthic species, for instance, may be under-represented; Beauchamp 1995), and fish removal may be biased towards larger fish (EPA 2000). Furthermore, depletion electroshocking with personnel increases in difficulty as the waterway increases in size. While it is easy to float downstream in a river, shocking a transect, it would be comparatively difficult to shock the entire width of the San Joaquin River in the Restoration Area. And while the scope of this paper focuses on predator removal at structures, merely removing predators at these locations would likely result in immediate re-colonization from open sections of the river (Kano 1987; Ledgerwood *et al.* 1994; Cavallo *et al.* 2012). Kano (1987) notes that control techniques would need to be frequent and extensive in order to affect a large portion of predators. On the other hand, intensive efforts could be concentrated around critical periods of fish movements (*e.g.,* juvenile salmon outmigration; Ledgerwood *et al.* 1994; Cavallo *et al.* 2012). Rather than eradication, long-term electrical-based efforts could be directed at suppressing populations of piscivorous fish (Peterson *et al.* 2008). The use of an electrical crowder, though, shows promise for fish control at facilities; while the development of the crowder described (Svoboda and Horn 2013) was designed for use within a screening facility (Tracy Fish Collection Facility, TFCF), future development could potentially be used to deter larger predatory fish from holding near structures where predator numbers are often greater (Rieman *et al.* 1991; Ward *et al.* 1995; Vogel *et al.* 1998).

Chemical Control

In a review of fish control projects, those involving the use of chemicals were most frequently cited as successful (Meronek *et al.* 1996). Often times, chemicals are applied by dosing a body of water, followed by some kind of mixing action (*e.g.,* using the outboard motor of a boat) to agitate the water and disperse the product (Krumholz 1948). Chemical control, as with any other fish control, typically aims at containment, management, or eradication of a target fish species (Clearwater *et al.* 2008).

Rotenone, an extract from plants of the Leguminosae family, has long been used by humans for gathering fish, and has been used in fisheries research since the early 20th century (Krumholz 1948). Generally applied in a powder or liquid application to water bodies, bait-based rotenone is also available (to target certain cyprinids; Rowe 2001). When applied on a widespread basis to water bodies, though, rotenone is generally nonspecific and will kill many or all fish species present (Rowe 2011). To an extent, larger fish are less susceptible to rotenone application and may remain alive longer than smaller fish (Chadderton *et al.* 2001). Rotenone can maintain potency from several days to beyond a month, depending on water conditions (*e.g.,* temperature, alkalinity, sunlight; Krumholz 1948; Chadderton *et al.* 2001). Rotenone was applied to five ponds in New Zealand (Chadderton *et al.* 2001); while concentrations significantly declined after 24 h, it took 15-50 days before rotenone was undetectable in these ponds. Often, potassium permanganate is used to detoxify rotenone (Lintermans and Raadik 2001). Rotenone affects not only fish but also aquatic invertebrates and some amphibians, causing death, but has little effect on other vertebrates (Krumholz 1948). Another method of using rotenone is to confine spawning grounds of fish using block nets, followed by dosing in the immediate area (Rowe 2001). This would require fish to be restricted to a rather small area, though, for it to be economical to dose; further, this would require the background knowledge to locate target fish spawning areas and require they generally be within the same vicinity for this technique to be effective.

Like rotenone, antimycin is a non-specific piscicide that inhibits cellular respiration and is used for eradication of nuisance species (Clearwater *et al.* 2008). Salmonids are typically more sensitive to antimycin than other fish species (Clearwater *et al.* 2008). Like rotenone, antimycin can be detoxified using potassium permanganate. Because antimycin is an antibiotic, long-term use may cause development of resistant strains of bacteria (Clearwater *et al.* 2008). Arnica Creek, a tributary of Yellowstone Lake, was treated with antimycin to remove a population of introduced brook trout thought to threaten a population of native Yellowstone cutthroat trout (*O. clarki bouvieri*; Gresswell 1991). Drip stations were used at selected points along the tributary to dose antimycin. Backpack sprayers were used in areas with insufficient flow to disperse the toxin, as well. During applications when Arnica Creek was connected to Yellowstone Lake, potassium permanganate was used to detoxify antimycin to prevent accidental dosing in the lake where native populations of cutthroat trout lived. While no brook trout were found in the tributary following dosing, reduced populations of cutthroat trout were described; which was an unavoidable consequence of using a non-specific piscicide. So, while antimycin was considered effective for removing fish, its use was confined to a tributary of a larger water body, and its dispersal into this water body was controlled to prevent unacceptable losses of other fish species (namely the native cutthroat trout).

Dry ice has been used for deoxygenating purposes to remove fish (Clearwater *et al.* 2008; Wu and Bridges, *in draft*). Its effect on fishes is variable, depending on a particular species' tolerance to low oxygen levels; salmonids are often considered to be more sensitive than other species (Clearwater *et al.* 2008). At the TFCF, dry ice was applied to bypass tubes which carry water from a primary diversion channel to a secondary channel, and ultimately, to holding tanks where fish are removed from the system (Wu and Bridges, *in draft*). Predatory fish, which are able to reside in flows in the bypass channels are thought to consume fish that enter the facility. Flows were reduced in the bypass channels and dry ice added to increase CO_2 concentrations. After anesthetization by CO_2 addition, flows in bypass tubes were increased and fish flushed into nets downstream.

While increasing CO_2 concentrations were found to be effective for removing fish from this holding facility, evaluations of re-colonization by predatory fish were not evaluated. Use of CO_2 in this situation was restricted to confined area (bypass tubes) where water flows could be controlled. In the Restoration Area, CO_2 would likely be restricted to similar situations. Water flows would need to be controlled to adequately dose CO_2 to target levels. If not, a greater quantity would be necessary, which could become prohibitively expensive. Also, fish were confined to bypass tubes and were unable to relocate to areas of low CO_2 concentration. At an open structure (*e.g.*, drum screen, dam face, and weir), fish would likely away these structures, preventing capture and removal.

A number of other piscicides are available (*e.g.*, copper sulfate, sodium sulphite, lime, bleaching powder, saponins); however, their use is generally limited to smaller water bodies, is not cost effective, is considered inhumane, or presents other environmental concerns that would prevent their use in riverine settings (Clearwater *et al.* 2008). Though not widely developed, the use of fish pheromones has also been introduced as a means to control fish populations (Sorensen and Stacey 2004; Britton *et al.* 2010). The use of pheromones has been suggested because they are often species specific and break down relatively quickly. A broad array of pheromones exist that can elicit different responses from fish, such a schooling (reproductive responses) or dispersal (anti-predatory responses), that could allow for capture or repelling target fish species. However, large-scale synthesis and use of pheromones is not currently available, particularly for piscivorous fish in the San Joaquin River.

Successful control with chemicals is usually restricted to small systems, such as mountain streams, or closed systems, such as lakes and ponds (Gresswell 1991; Meronek *et al.* 1996; Mueller 2005). Eradication or reduction of nuisance species in these areas are possible because the size and isolation of these systems. Smaller system size is important because it allows for target concentrations of chemicals to be reached, which could be difficult (and cost-prohibitive) in larger systems, particularly flowing systems where a dilution from flowing waters would also exist (Rowe 2001). Isolation of systems typically treated with chemicals is another important factor because isolated systems prevent re-colonization by species targeted for removal. Without isolation, nuisance species would likely return to treated areas, requiring long-term use of chemicals for effective control. While sections of the Restoration Area de-water during low-flow seasons, successful use of chemicals, particularly at in-river structures would likely not be feasible. Removal would likely be short-lived, too, as re-colonization by predators would occur when increasing flows re-connect sections of the river still harboring predators.

In addition to the feasibility of chemical use in a riverine system, strong public opposition would likely preclude chemical use in the Restoration Area. When plans to use toxins to remove northern pike (*Esox lucius)* from Lake Davis were brought to public attention, residents quickly opposed California Department of

Fish and Game's (DFG) proposal (Goedde 1998). While DFG ultimately decided to follow through with these plans, a lengthy economic cost/benefit analysis, extensive plans to prevent release of toxins outside of Lake Davis and into drinking water supplies, and even lawsuits by residents against DFG ensued. While any control method will have a public and environmental safety component, it is even more prudent when dealing with chemicals (Clearwater *et al.* 2008). Public opposition to the use of chemicals for controlling populations of organisms is not unwarranted, though. Significant associations to pesticide use and diseases, such as Parkinson's disease have been demonstrated (Tanner *et al.* 2011).

If chemicals are used to remove fish from a waterway, disposal afterwards is another issue that requires attention (Krumholz 1948). Nutrient balances could be disrupted from large fish kills, which could have detrimental effects on an ecosystem (Clearwater *et al.* 2008). Furthermore, because non-target species would also be affected by most chemicals, this would mean that native fish, including any threatened and endangered species, would be at risk for eradication, in addition to targeted fish species (Rowe 2001). Because of issues regarding environmental and safety concerns, chemical use in the Restoration Area is likely not the best option for predator removal at structures.

Discussion

There are inherent trade-offs with any of the control methods described above. While chemical control may be the most direct, and remove all fish in the immediate area of its application, it is also more difficult to dose in a larger system, particularly a flowing system. Unlike mechanical methods, where only the immediate area is affected, chemicals would be distributed over a broader area, potentially causing safety concerns to downstream water users, and would likely have greater push-back from the general public. Mechanical and electrical control methods, on the other hand, are easier to control and target certain fish. While multiple species may be captured or removed with these methods, non-target fish could be returned to the river. However, these qualities make mechanical and electrical control more labor intensive. Nonetheless, a combination of methods (mechanical and electrical) is most likely to provide positive results when controlling piscivorous fish at in-river structures. Electrical crowders have shown promise at deterring larger fish from holding near structures (Svoboda and Horn 2013), and electrical arrays have been used to remove piscivores within screening facilities (Kano 1987). The use of nets to exclude larger fish from areas where agencies want to control access has been demonstrated (Knapp and Matthews 1998). A combination of these methods could be utilized within the Restoration Area to temporarily control predators near structures, particularly during critical periods when native species may be migrating near such structures.

A shortfall of most predator control studies is that efforts are only immediately recognized. Few studies evaluate the results of control efforts over extended periods of time. Reported results of these studies typically describe the numbers or biomass of predators removed (Liston *et al.* 1994) or increases in prey-fish recovery rates immediately following predator removals (Kano 1987; Ledgerwood *et al.* 1994; Cavallo *et al.* 2012). In other studies, expectations of control efforts may be modeled (Beamesderfer *et al.* 1996; Peterson *et al.* 2008). Each waterway, where predator control efforts are conducted, is going to have a unique set of issues that affect predation and survival of resident fish. However, this does not preclude the use of predator control methods in the Restoration Area. Even though long-term increases in fish survival may be difficult to predict, removal efforts, nonetheless, have the potential to yield positive short-term results.

Beyond the sustainability of predator control efforts, other problems from predator control programs could surface. The San Joaquin River is largely dominated by nonnative fish (SJRRP 2013, *in draft*). Removing nonnative predators in an attempt to promote native fish recruitment could cause a compensatory response by other nonnative fish (Zavaleta *et al.* 2001). In ecosystems with numerous introduced species, results of targeting only one nonnative species for removal or control can have unintended results. For example, removal of largemouth bass in an ephemeral pond on Lake Mohave was coincident with increased abundance of crayfish and odonate (dragonfly) larvae which potentially caused an increase in predation on larval fish (Mueller and Burke 2005). In another pond from this study, largemouth bass removal was concurrent with an increase in centrarchids (+285%). Schooling razorback suckers in the area were observed with wounds on fins and caudal peduncles, being harassed by sunfish. After rotenone was applied a section of the North Fork of the Feather River in California, it was noted that the reduction in pikeminnow populations may have allowed a population of smallmouth bass to establish (Moyle *et al.* 1983). During removal of smallmouth bass in the Yampa River, nonnative small-bodied fish recruitment increased, possibly due to reduced predation pressure (Bestgen *et al.* 2007).

Nearly all predator control methods have some sort of size bias, generally skewed towards larger fish (Larson *et al.* 1986; Knapp and Matthews 1998; Chadderton *et al.* 2001). While some species have a positive correlation between size and salmonid predation (pikeminnow, catfish; Poe *et al.* 1991; Vigg *et al.* 1991), others have a negative correlation (black bass; Vigg *et al.* 1991; Tabor *et al.* 2007). Responses in systems with native and nonnative species to predator control efforts should be understood prior to large-scale efforts. Without this knowledge, potentially detrimental results could occur from targeting single species for removal.

INDIRECT CONTROL

“[H]abitat improvements are usually more appropriate than fish population manipulations where introduced species thrive in altered habitats poorly suited for the native species (Beamesderfer 2000).” Because of environmental conditions in Central Valley streams, sustained predator removal, while possible, may not be practical because river habitat favors nonnative species over native ones (Marchetti and Moyle 1991; Reyenolds *et al.* 1993). Predator reductions from direct-control methods are usually short-lived because predators re-colonize these areas shortly after removal (Kano 1987; Cavallo *et al.* 2012). This is an inherent problem of an open system; there are no restrictions to prevent predators in other areas of the river from re-colonizing after removals (Gingras and McGee 1997).

Predator-control methods that require physical removal of predators have been largely dismissed in the scientific community (Mueller 2005; Beamesderfer 2000). Methods are often labor intensive and results describing the direct success of these programs are limited. While predation on juvenile salmon may be detrimental to the species’ survival, the real issue is more likely that available habitat is unsuitable and favors nonnative fish. Long-term goals should focus on addressing the cause of the problem, rather than the symptoms (Meronek *et al.* 1996). While direct predator control efforts may initially reduce predation, continuing these efforts long-term may prove difficult. Intensive, sustained predator removal efforts may be necessary before increases in survival of target fish are realized (Cavallo *et al.* 2012). Even then, it is difficult to predict the outcome of predator control efforts. Only after significant involvement has been undertaken can results of these efforts be realized. While predator control could function to facilitate salmon reintroductions early in the restoration process, long-term efforts should shift towards sustainable practices that more effectively balance requirements of native species with river operations. An in-depth discussion of long-term goals that facilitate restoration efforts and reduce nonnative predation is beyond the scope of this paper. However, some key components are discussed in the following sections.

Facility Improvements

Structures such as dams and screening facilities provide refuge for predators, in addition to an influx of prey (Kano 1987; Rieman *et al.* 1991, BOR 2006). Facility operations can be adjusted to include regular predator removal efforts. However, like many direct control methods, this merely addresses symptoms of a larger problem- the problem of many screening facilities being that the construction design provides piscivorous fish with low velocity refuges and concentrate prey fish, funneling them towards these predators.

The TFCF is a fish salvage facility that diverts fish that would otherwise be entrained by downstream pumping facilities (*i.e.,* C.W. “Bill” Jones Pumping Plant that diverts water to the Delta-Mendota Canal; Liston *et al.* 1994). Striped bass and white catfish congregate in areas upstream and within the TFCF. In the TFCF, a series of louvers and bypass channels divert fish into holding tanks where fish are collected and transported downstream in the Delta. Predatory fish within the facility prey on smaller fish as they pass through the facility. Because threatened and endangered species enter the facility, regular predator control efforts take place at the TFCF (Liston *et al.* 1994). These efforts included lowering water in the secondary channel and flushing of the bypass tubes, in an effort to remove fish holding against the water velocities.

The Red Bluff Diversion Dam is a low-head dam, located near Red Bluff, California. The facility diverts water from the Sacramento River to the Corning and Tehama-Colusa Canals. The low-head dam creates a predator holding area because of slack flows behind the structure, where striped bass and pikeminnow congregate (Tucker *et al.* 1998). Juvenile salmon passing through the facility were noticeable disoriented, and those straying higher into the water column were often consumed by pikeminnow (Vogel *et al.* 1988). In 1984, juvenile salmonid loss by predation to pikeminnow was attributed as the primary loss (16–55%) of juvenile fish at this facility. In an attempt to reduce predator abundance near the dam, the dam gates were raised (starting in 1986) during non-irrigation seasons to allow dispersion and upstream pikeminnow movement. A reduction in pikeminnow holding below the dam was noted following this practice.

Beyond adjusting facility operations to support predator reductions, designing and improving facilities to minimize predator holding addresses a main problem regarding predation at in-river structures. While future structures are an integral component of the restoration process to prevent fish entrainment losses, the very nature of their design provides a velocity refuge for predators, and a constant source of prey (Kano 1987; Vogel *et al.* 1988; Vigg *et al.* 1991). Fish passing through these facilities are often weak and disoriented; directing fish through facilities typically concentrates them as well, providing an easy source of food for predators holding in velocity refuges (BOR 2006). Facilities should be designed, or modified, to minimize these low velocity refuges (Kano 1987; BOR 2006; Odenweller, *pers. comm.*). Fish outfalls should be designed to exit where predators may not hold as easily (high-velocity areas), and should not concentrate fish in areas easily accessible to predators (BOR 2006).

Regulatory Changes/Public Recruitment

While agency-based angling may be less effective than other direct control methods (Liston *et al.* 1994), public participation in angling efforts has the potential to reduce piscivores throughout the Restoration Area. Public engagement can elicit response from a larger group of people than would otherwise be available using agency personnel. Methods to encourage public participation include removal of regulatory restrictions of fish take (Tyus and Saunders 2000), opening waterways to fishing that would otherwise be restricted (Larson *et al.* 1986; Paul *et al.* 2003), or development of a sport-reward fishery that provides anglers compensation for removal of fish targeted for exploitation (Zimmerman *et al.* 2000; Porter 2011).

As an alternative to the high cost of labor and equipment of electroshocking efforts, an experimental fishery was evaluated for removal of rainbow trout in a stream in Great Smokey Mountains National Park (Larson *et al.* 1986). While angler participation was initially high, it dropped rapidly after the first two weeks of a nine-week program. About half of the fish removed during this time were also removed during these first two weeks. However, catch-per-unit-effort was greater for angler-based efforts when compared to electrofishing efforts. It was noted that fish < 100 mm were not well represented in angler take, though. Because angler efforts declined with time, employing this method alone could fail to achieve target exploitation rates, and efforts may need to be supplemented. A similar study was evaluated at Quirk Creek, Alberta (Paul *et al.* 2003). In an attempt to remove nonnative brook trout threatening native bull (*S. confluentus*) and cutthroat trout, groups of anglers were taken on organized trips to the study area. Anglers first had to pass a fish-identification test to ensure proper harvest of target species. Angling efforts took place over a 3-yr. period, from 1998 to 2000. Angling had little effect on brook trout populations. Because of their fast growth rate, low catchability, and early maturation, they were resilient to exploitation efforts. Conversely, native species had a higher catchability rate. Combined with incidental take (*i.e.,* mortality from hooking), selective harvest in this ecosystem had the potential to be detrimental to native fish populations.

One of the most extensive predator removal efforts, the pikeminnow sport-reward program, takes place in the Columbia River Basin (Beamesderfer *et al.* 1996; Friesen and Ward 1999; Porter 2011). In 2011, efforts included a sport-reward fisheries program that awarded anglers for harvesting pikeminnow >228 mm (9 inches) in a tiered system, rewarding greater monetary value for increasing numbers of fish, as well as rewards for returning spaghetti-tagged (Floy Tag, Inc., Seattle, WA) pikeminnow (Porter 2011); the basis of the program awarded anglers $4 per pikeminnow returned to a registration station, for the first 100 fish caught. Increasing amounts were available for anglers returning >100 fish to the station ($5 for fish 101–400, $8 for fish >400). Anglers were awarded $500 for spaghetti-tagged pikeminnow returned to the registration station. The spaghetti-tagged fish were used in conjunction with final numbers of harvested pikeminnow, to calculate annual exploitation rates. Pikeminnow returned under the sport-reward program in 2011 totaled 155,312. Total payments in 2011, under this program, totaled $1,062,188. System wide pikeminnow exploitation efforts suggest that a 15% pikeminnow exploitation rate was achieved which resulted in an estimated 36% reduction in salmonid predation.

Directly measuring reduction in predation of fishes proves difficult because of the complexity of the Columbia River system. Assumptions used in calculating the reduction in predation include: no compensatory response by other piscivorous fish, no variation in returning salmon adults or juvenile survival, no changes in river flows, and no changes in dam operations and turbine mortality (Beamesderfer *et al.* 1990). Because of these variables, it is difficult to directly measure the success of pikeminnow exploitation efforts (Beamesderfer *et al.* 1996; Friesen and Ward 1999).

Friesen and Ward (1999) state that any predator reduction program should only be one aspect of any management plan. For the San Joaquin Restoration Program, this means that predation at structures should be evaluated on the basis of the whole Restoration Reach before engaging in predator control efforts directed only at in-river structures (*e.g.,* Chowchilla Bifurcation Structure, Mendota Dam, Sack Dam/Arroyo Canal). While predation may be higher at these structures than other sections of the river, the sections of river without structures encompass a larger area, and a greater number of fish may be lost to predation in the stretches of river without man-made structures. For example, Rieman *et al.* (1991) states that predation "was more than 50 times greater immediately below the dam than that in the remaining reservoir," though predation outside the restricted zone below the dam accounted for 79% of the total predation in the system. Predatory fish are not exclusive to structures alone, and are present throughout the Restoration Area (SJRRP 2013, *in draft*). Focusing on predator control at structures alone may neglect the issue that predation in the remaining stretches of river may, by far, exceed predation at these structures.

Public angling may not directly reduce predator numbers near in-river structures, but long-term exploitation could reduce predator numbers in the Restoration Area, reducing predation pressure overall. Reducing predation at structures alone may not sufficiently alleviate predation pressure on juvenile salmon in the San Joaquin River; public participation by angling may help in reducing problematic predators (Beamesderfer *et al.* 1990). Tyus and Saunders (2000) suggest removal of take limits could encourage public angling and removal of target fish. However, restrictions exist only for striped bass (daily limit=2, min size=46 cm), black bass (daily limit=5), and sunfish (including crappie, daily limit=25) for nonnative piscivorous fish in the Restoration Area (DFW Regulations 2013). While important to determine which size class of fish need to be targeted for exploitation to maximize reduction in predation on native fish, take limits could be lifted in the Restoration Area to encourage public involvement and removal of these fish. For those predators identified as most problematic, a similar sport-reward fishing program could take place in the Restoration Area.

Restoration-based Efforts

Because of the potential for a constant influx of predators to in-river structures from other sections of river, efforts to reduce predators throughout the Restoration Area should be addressed. Successful establishment by nonnative fishes often correlates with environmental degradation (Britton *et al.* 2010). A growing field of evidence supports this statement (Brown 2000; Marchetti and Moyle 2001; Feyrer and Healy 2003). The mere introduction of alien species does not guarantee that species will become resident in the introduced system (Moyle and Light 1996). However, the gross alteration of available habitat and flows in the San Joaquin River support introduced warm-water species, and has disfavored native fish distribution in most of the Restoration Area.

Over a 5-yr period, fish assemblages were monitored in Putah Creek (Marchetti and Moyle 2001). During this time, a number of unusually dry years were followed by a number of unusually wet years. Nonnative fish abundance decreased in response to increasing flow, typically in downstream reaches where nonnative fish were usually dominant. It is hypothesized that higher summer flows provide cooler water to downstream reaches while reducing the favorability of habitat to nonnative fishes. Not only do higher flows increase native fish distribution by providing reduced temperatures, pool habitat, and water conductivity, but they also increase travel time of downstream-migrating fish (Marchetti and Moyle 2001; Cavallo *et al.* 2012). It is also thought that decreased temperatures from increasing flows decrease nonnative predator metabolism and increase salmonid survival. Because the Restoration Area is the southernmost range for Chinook salmon, increased temperatures from reduced flows increases mortality of these fish (Collis *et al.* 1995; Marine and Cech 2004). Dam releases providing continuous flow through the Restoration Area could help to reduce temperatures and increase flow in pools that may harbor nonnative predators. Increasing flows can reduce juvenile salmon outmigration time, reducing temporal exposure to predation, and promoting increased survival (Raymond 1979; Kondolf *et al.* 1996; Cavallo *et al.* 2012).

Other than the issue of insufficient river flows, several gravel pits exist in the Restoration Area. While abandoned mine pits are not an in-river structure, *per se,* the removal of streambed material creates an unnatural habitat that presents unique obstacles for native fish that would not otherwise be present (Reyenolds *et al.* 1993; Kondolf *et al.* 1996). These gravel pits have created areas where river flows reduce to slack water, which can disorient outmigrating smolts and provide habitat for black bass (Kondolf *et al.* 1996). Additionally, water temperatures in these sections of river may rise because otherwise flowing water slows and is exposed to sunlight for long periods of time (Norman *et al.* 1998).

From 1996−1997, sea-migrating brown trout (*Salmo trutta*) in Sweden were tracked using passive integrative transponder (PIT) tags to determine loss through an in-river pond on Önnerupsbäcken Creek (Olsson *et al.* 2001). Clay-pit ponds have been joined with rivers and streams in Sweden for denitrification of agricultural runoff. However, the issues that resulted from this in-river structure were initially overlooked. While the migrating fish species, brown trout, and the primary resident predator, northern pike, are not the target species in the San Joaquin River, this study presents similar issues that exist with the in-river mine pits in the Restoration Area. Results suggest an increased time for smolts migrating through the pond and greater migration losses, compared to stream sections. This loss increase was suggested to be a factor of predation by resident predators and loss of migration cues, such as reduced flow rates through the pond section.

Similar in structure to the denitrification ponds used on the Önnerupsbäcken Creek, several gravel extraction pits exist on the Tuolumne River in California (McBain & Trush 2000). Along with other compounding factors, these pits were determined to be a refuge for predators that would ultimately reduce outmigrating salmonids (EA Engineering 1992). In 2001, Special Run Pool 9 (SRP 9), a wide pool on the Tuolumne River, created by in-stream gravel mining, was reconstructed, filling in the majority of the gravel pit to more closely resemble the original river channel (TID and MID 2005). While the reconstruction was successful, the restoration effort was ultimately determined unsuccessful at reducing nonnative predators. Black bass populations decreased following a 1997 flood but increased during low flow years thereafter; 2003 predator estimates did not reveal a significant reduction in predator populations (Stillwater 2006). However, the idea has been proposed that the failure to reduce predators doesn’t necessarily equate in a failure to reduce predation. It has been suggested that black bass hold closer to river margins while salmon migrate through more central portions of the channel, effectively providing spatial separation between predator and prey (TID and MID 2005). River 2D modeling of SRP 9 suggests that normal spring flows can provide the velocity gradient to provide a corridor for outmigrating salmonids from black bass (McBain and Trush 2006).

CONCLUSION

Unfortunately, information regarding the success of predator removal programs is far from conclusive. Beamesderfer (2000) highlights three questions when deciding to pursue programs such as predator control: Is the identified problem significant? Can it be affected? Is it acceptable? Often, studies fail to acknowledge whether programs are sustainable, or can remove sufficient numbers of predators to increase native fish recruitment. Post-evaluation of any control program is necessary to provide valuable information to the program. However, few studies evaluate the relevant issue: predatory fish are removed to help a target species (*e.g.,* Chinook salmon). So, after predator removal, did numbers of the target species increase (Novinger and Rahel 2003)? Is recruitment now higher than before (Mueller 2005)?

While direct control efforts are not likely sustainable, they should be included as part of broader restoration efforts. During initial efforts, direct control may facilitate reintroduction of salmon in the Restoration Area by reducing immediate predation pressure. However, long-term efforts should be directed at increasing habitat suitable for native fish while concurrently reducing habitat that favors nonnative species. Unfortunately, there is no simple solution to a complex problem. Merely removing predators at structures may fail to alleviate pressure from predation on fishes. Because the San Joaquin River is essentially an open system, recruitment at in-river structures may prove to be too great to realize benefits from removal efforts. On the other hand, predator removal efforts may be effective for reducing predation pressure if efforts are concentrated during periods of fish passage (*i.e.,* salmonid outmigration).

"[H]abitat restoration efforts should be evaluated *a priori* and biologically prioritized so that scarce resources can be allocated to efforts with the greatest potential and least amount of risk, in terms of meeting conservation and recovery goals" (Budy and Schaller 2007). Prior to committing funds to a predator control program, it should be evaluated whether or not the limiting factor is piscivorous fish (Kondolf *et al.* 2008). If so, a programmatic approach that will attempt to alleviate predation by long-term reduction in predators, whereby the river provides habitat that was once favorable to salmon while potentially reducing habitat suitable to nonnative predators (along with other nonnative fish), should be considered. Merely attempting to remove piscivorous fish without addressing the issues that provide for their recruitment may result in a failure to achieve target restoration goals.

ACKNOWLEDGMENTS

I would like to thank Don Portz and Zak Sutphin, Bureau of Reclamation, for providing invaluable guidance while completing this research. I would also like to thank Dan Odenweller, California Department of Fish and Game, retired, Joe Merz, Cramer Fish Sciences, and Jose Setka, East Bay Municipal Utility District, for taking time from their busy schedules to discuss predation issues with me, while providing direction and valuable fisheries knowledge from their own research and expertise.

LITERATURE CITED

Beamesderfer, R.C.P. 2000. Managing fish predators and competitors: Deciding when intervention is effective and appropriate. Fisheries Management 25: 18–23.

Beamesderfer, R.C.P., B.E. Rieman, L.J. Bledsoe, and S. Vigg. 1990. Management implications of a model of predation by a resident fish on juvenile salmonids migrating through a Columbia River reservoir. North American Journal of Fisheries Management 10: 290–304.

Beamesderfer, R.C.P., D.L. Ward, and A.N. Nigro. 1996. Evaluation of the biological basis for a predator control program on northern squawfish (*Ptychocheilus oregonensis*) in the Columbia and Snake Rivers. Canadian Journal of Fisheries and Aquatic Sciences 53: 2898–2908.

Beauchamp, D.A. 1990. Seasonal and diel food habits of rainbow trout stocked as juveniles in Lake Washington. Transactions of the American Fisheries Society 119: 475–482.

Beauchamp, D.A. 1995. Riverine predation on sockeye salmon fry migrating to Lake Washington. North American Journal of Fisheries Management 15: 358–365.

Bestgen, K.R., C.D. Walford, A.A. Hill, and J.A. Hawkins. 2007. Native fish response to removal of non-native predator fish in the Yampa River, Colorado. Colorado River Recovery Implementation Program Project Number 140. Larval Fish Laboratory Contribution 150.

[BOR] Bureau of Reclamation. 2006. Fish Protection at Water Diversions. Denver; Water Resources Technical Publication. 480 p.

Britton, J.R., R.E. Gozlan, and G.H. Copp. 2010. Managing non-native fish in the environment. Fish and Fisheries 12: 256–274.

Brown, L.R. 2000. Fish communities and their associations with environmental variables, lower San Joaquin River drainage, California. Environmental Biology of Fishes 57: 251–269.

Brown, L.R. and D. Michniuk. 2007. Littoral fish assemblages of the alien-dominated Sacramento-San Joaquin Delta, California, 1980-1983 and 2001-2003. Estuaries and Coasts 30: 186–200.

Browne, D.R. and J.B. Rasmussen. 2009. Shifts in the trophic ecology of brook trout resulting from interactions with yellow perch: an intraguild predator-prey interaction. Transactions of the American Fisheries Society 138: 1109–1122.

Budy, P. and H. Schaller. 2007. Evaluating tributary restoration potential for Pacific salmon recovery. Ecological Applications 17: 1068–1086.

Cavallo, B., J. Merz, and J. Setka. 2012. Effects of predator and flow manipulation on Chinook salmon (*Oncorhynchus tshawytscha*) survival in an imperiled estuary. Environmental Biology of Fishes (2012): 1–11.

Chadderton, L., S. Kelleher, A. Brow, T. Shaw, B. Studholme, and R. Barrier. 2001. Testing the efficacy of rotenone as a piscicide for New Zealand pest species. Managing Invasive Freshwater Fish in New Zealand. Proceedings of a workshop hosted by the Department of Conservation, pp. 131–142.

Clearwater, S.J., C.W. Hickey, and M.L. Martin. 2008. Overview of potential piscicides and molluscicides for controlling aquatic pest species in New Zealand. Science for Conservation 283. 74 p.

Collis, K., R.E. Beaty, and B.R. Crain. 1995. Changes in catch rate and diet of northern squawfish associated with the release of hatchery-reared juvenile salmonids in a Columbia River reservoir. North American Journal of Fisheries Management 15: 346–357.

[DFW Regulations] 2013-2014 Freshwater Sport Fishing Regulations. 2013. California Department of Fish and Wildlife, Sacramento, CA.

Dudley, R.K. and W.J. Matter. 2000. Effects of small green sunfish (*Lepomis cyanellus*) on recruitment of Gila chub (*Gila intermedia*) in Sabino Creek, Arizona. The Southwestern Naturalist 45: 24–29.

Dunham, J.B., S.B. Adams, R.E. Schroeter, and D.C. Novinger. 2002. Alien invasions in aquatic ecosystems: toward an understanding of brook trout invasions and potential impacts on inland cutthroat trout in western North America. Reviews in Fish Biology and Fisheries 12: 373–391.

[EA Engineering] EA Engineering, Science, and Technology. 1992. Fisheries studies report. In Report of the Turlock Irrigation District and Modesto Irrigation District pursuant to Article 39 of the license for the Don Pedro project (Project No. 2299) before the US Federal Energy Regulatory Commission (Vol. II).

[EPA] United States Environmental Protection Agency. 2000. New perspectives in electroshocking. Office of Research and Development, EPA/600/R-99/108.

East, P. and P. Magnan. 1991. Some factors regulating piscivory of brook trout, *Salvelinus fontinalis*, in lakes of the Laurentian Shield. Canadian Journal of Fisheries and Aquatic Sciences 48: 1735–1743.

Feyrer, F. and M.P. Healey. 2003. Fish community structure and environmental correlates in the highly altered southern Sacramento-San Joaquin Delta. Environmental Biology of Fishes 66: 123–132.

Friesen, T.A. and D.L. Ward. 1999. Management of northern pikeminnow and implications for juvenile salmonid survival in the Lower Columbia and Snake Rivers. North American Journal of Fisheries Management 19: 406–420.

Fritts, A.L. and T.N. Pearsons. 2004. Smallmouth bass predation on hatchery and wild salmonids in the Yakima River, Washington. Transactions of the American Fisheries Society 133: 880–895.

Gingras, M. and M. McGee. 1997. A telemetry study of striped bass emigration from Clifton Court Forebay: Implications for predator enumeration and control. California Department of Fish and Game, Technical Report 54.

Goedde, N.R. 1998. The poisoning of Lake Davis: Weighing the risks. Environs: Environmental Law and Policy Journal 21: 3–24.

Gresswell, R.E. 1991. Use of antimycin for removal of brook trout from a tributary of Yellowstone Lake. North American Journal of Fisheries Management 11: 83–90.

Hall, Jr., F.A. 1979. An evaluation of downstream migrant Chinook salmon (*Oncorhynchus tshawytscha*) losses at Hallwood-Cordua fish screen. Anadromous Fisheries Branch Administrative Report No. 79-5.

Kano, R.M. 1987. The effects of trashrack and bypass design and predator control on predation losses of juvenile Chinook salmon at Hallwood-Cordua fish screen. Interagency Ecological Study Program for the Sacramento-San Joaquin Estuary.

Kim, G.W. and D.R. DeVries. 2001. Adult fish predation on freshwater limnetic fish larvae: a mesocosm experiment. Transactions of the American Fisheries Society, 130: 189–203.

Knapp, R.A. and K.R. Matthews. 1998. Eradication of nonnative fish by gill netting from a small mountain lake in California. Restoration Ecology 6: 207–213.

Kondolf, G.M., P.L. Angermeier, K. Cummins, T. Dunne, M. Healey, W. Kimmerer, P.B. Moyle, D. Murphy, D. Patten, S. Railsback, D.J. Reed, R. Spies, and R. Twiss. 2008. Projecting cumulative benefits of multiple river restoration projects: An example from the Sacramento-San Joaquin River System in California. Environmental Management 42: 933–945.

Kondolf, G.M., J.C. Vick, and T.M. Ramirez. 1996. Salmon spawning habitat rehabilitation on the Merced River, California: an evaluation of project planning and performance. Transactions of the American Fisheries Society 125: 899–912.

Krueger, D.M., E.S. Rutherford, and D.M. Mason. 2011. Influence of predation mortality on survival of Chinook salmon parr in a Lake Michigan tributary. Transaction of the American Fisheries Society 140: 147–163.

Krumholz, L.A. 1948. The use of rotenone in fisheries research. The Journal of Wildlife Management 12: 305–317.

Larson, G.L., S.E. Moore, and D.C. Lee. 1986. Angling and electrofishing for removing nonnative rainbow trout from a stream in a national park. North American Journal of Fisheries Management 6: 580–585.

Ledgerwood, R.D., E.M. Dawley, L.G. Gilbreath, M.A. Toner, T.P. Poe, and H.L. Hansen. 1994. Effectiveness of predator removal for protecting juvenile fall Chinook salmon released from Bonneville Hatchery, 1993. Report by National Marine Fisheries Service to the U.S. Department of Energy, Bonneville Power Administration, Division of Wildlife, Seattle, Washington, Project No. 90-077, 57 p.

Lintermans, M. and T. Raadik. 2001. Local eradication of trout from streams using rotenone: the Australian experience. Managing Invasive Freshwater Fish in New Zealand. Proceedings of a workshop hosted by the Department of Conservation, pp. 95–111.

Liston, C., C. Karp, L. Hess, and S. Heibert. 1994. Summary of the fish predator removal program and intake channel studies, 1991–1992. Tracy Fish Collection Facility Studies, California, Volume 1. U.S. Department of the Interior, Bureau of Reclamation Mid-Pacific Region and Denver Technical Services Center.

Marchetti, M.P. and P.B. Moyle. 2001. Effects of flow regime on fish assemblages in a regulated California stream. Ecological Applications 11: 530–539.

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.

McBain & Trush. 2000. Habitat restoration plan for the lower Tuolumne River corridor. Final report to the Tuolumne River Technical Advisory Committee, McBain and Trush, Arcata, California.

Meronek, T.G., P.M. Bouchard, E.R. Buckner, T.M. Burri, K.K. Demmerly, D.C. Hateli, R.A. Klumb, S.H. Schmidt, and D.W. Coble. 1996. A review of fish control projects. North American Journal of Fisheries Management 16: 63–74.

Merz, J.E. 2002. Comparison of diets of prickly sculpin and juvenile fall-run Chinook salmon in the lower Mokelumne River, California. The Southwestern Naturalist 47: 195–204.

Moyle, P.B. 1973. Recent changes in the fish fauna of the San Joaquin River system. Cal-Nevada Wildlife 1973, 60–63.

Moyle, P.B. 2002. Inland Fishes of California. University of California Press.

Moyle, P.B and T. Light. 1996. Fish invasions in California: Do abiotic factors determine success? Ecology 77: 1666–1670.

Moyle, P.B., B. Vondracek, and G.D. Grossman. 1983. Responses of fish populations in the North Fork of the Feather River, California, to treatments with fish toxicants. North American Journal of Fisheries Management 3: 48–60.

Mueller, G.A. 2005. Predatory fish removal and native fish recovery in the Colorado River mainstem. Fisheries 30: 10–19.

Mueller, G.A. and T. Burke. 2005. Survival of young razorback sucker in relation to stocking rates (fish/ha) and in the presence or absence of predator communities in Lake Mohave, Arizona-Nevada. In: Proceedings from two symposia: Restoring Native Fish to the Lower Colorado River: Interactions of Native and Non-Native Fish and Restoring Natural Function within a Modified Riverine Environment: the Lower Colorado River. Albuquerque, NM: US Fish and Wildlife Service, Southwest Region. pp. 155–163.

Nobriga, M.L. and F. Feyrer. 2007. Shallow-water piscivore-prey dynamics in California’s Sacramento-San Joaquin Delta. San Francisco Estuary and Watershed Science 5(2).

Nobriga, M.L., F. Feyrer, and R.D. Baxter. 2006. Aspects of Sacramento pikeminnow biology in nearshore habitats of the Sacramento-San Joaquin Delta, California. Western North American Naturalist 66: 106–114.

Norman, D.K., C.J. Cederholm, and W.S. Lingley, Jr. 1998. Flood plains, salmon habitat, and sand and gravel mining. Washington Geology 26: 3–20.

Novinger, D.C. and F.J. Rahel. 2003. Isolation management with artificial barriers as a conservation strategy for cutthroat trout in headwater streams. Conservation Biology 17: 772–781.

[NRDC] Natural Resources Defense Council, *et al.*, v. Kirk Rodgers, as Regional Director of the United States Bureau of Reclamation, *et al.* 2006. Notice of filing of memorandum of understanding between settling parties and State of California. United States District Court, Eastern District of California. Case No. CIV S-88-1658 LKK/GGH.

Olsson, I.C., L.A. Greenberg, and A.G. Eklöv. 2001. Effect of an artificial pond on migrating brown trout smolts. North American Journal of Fisheries Management 21: 498–506.

Paul, A.J., J.R. Post, and J.D. Stelfox. 2003. Can anglers influence the abundance of native and nonnative salmonids in a stream from the Canadian Rocky Mountains? North American Journal of Fisheries Management 23: 109–119.

Peterson, D.P., K.D. Fausch, J. Watmough, and R.A. Cunjak. 2008. When eradication is not an option: modeling strategies for electrofishing suppression of nonnative brook trout to foster persistence of sympatric native cutthroat trout in small streams. North American Journal of Fisheries Management 28: 1847–1867.

Poe, T.P., H.C. Hansel, S. Vigg, D.E. Palmer, and L.A. Predergast. 1991. Feeding of predaceous fishes on out-migrating juvenile salmonids in John Day Reservoir, Columbia River. Transactions of the American Fisheries Society 120: 405–420.

Porter, R. 2011. Report on the predation index, predator control fisheries, and program evaluation for the Columbia River Basin experimental northern pikeminnow management program; 2011 annual report. Pacific States Marine Fisheries Commission, Project Number 1990-077-00.

Portz, D.P., N. Ponferrada, E. Best, and C. Hueth. 2012. Central Valley steelhead monitoring plan for the San Joaquin River Restoration Area. San Joaquin River Restoration Program. National Marine Fisheries Service Permit 16608 Report.

Raymond, H.L. 1979. Effects of dams and impoundments on migrations of juvenile Chinook salmon and steelhead from the Snake River, 1966 to 1975. Transactions of the American Fisheries Society 108: 505–529.

Reimers, R. 1955. Conditions of existence, growth, and longevity of brook trout in a small, high-altitude lake of the eastern Sierra Nevada. California Fish and Game, Vol. 44, No. 4.

Reynolds, F.L., T.J. Mills, R. Benthin , and A. Low. 1993. Restoring Central Valley: A plan for action. California Department of Fish and Game.

Rieman, B.E., R.C. Beamesderfer, S. Vigg, and T.P. Poe. 1991. Estimated loss of juvenile salmonids to predation by northern squawfish, walleyes, and smallmouth bass in John Day Reservoir, Columbia River. Transactions of the American Fisheries Society 120: 448–458.

Rowe, D.K. 2001. Rotenone-based approaches to pest fish control in New Zealand. Managing Invasive Freshwater Fish in New Zealand. Proceedings of a workshop hosted by the Department of Conservation, pp. 131–142.

Shepard, B.B., R. Spoon, and L. Nelson. 2002. A native westslope cutthroat trout population responds positively after brook trout removal and habitat restoration. Intermountain Journal of Sciences 8: 193–214.

Sorensen, P.W. and N.E. Stacey. 2004. Brief review of fish pheromones and discussion of their possible uses in the control of non-indigenous teleost fishes. New Zealand Journal of Marine and Freshwater Research 38: 399–417.

[SJRRP 2013] San Joaquin River Restoration Program. *In draft*. 2013 Inventory and Monitoring: San Joaquin River Restoration Area, *in draft*. San Joaquin River Restoration Program, Bureau of Reclamation, Sacramento, CA.

Snyder, J.O. 1940. The trouts of California. California State Print. Office.

[Stillwater] Stillwater Sciences and McBain & Trush, Inc. 2006. Lower Tuolumne river predation assessment. Final report to the Tuolumne River Technical Advisory Committee. Stillwater Sciences, Berkely, CA. McBain & Trush, Inc., Arcata, CA.

Svoboda, C. and M. Horn. 2013. Laboratory design and testing of an electrical crowder for predation reduction at the Tracy Fish Collection Facility. Bureau of Reclamation, Tracy Series, Vol. 48.

Tabor, R.A., G.S. Brown, and V.T. Luiting. 2004a. The effect of light intensity on sockeye salmon fry migratory behavior and predation by cottids in the Cedar River, Washington. North American Journal of Fisheries Management 24: 128–145.

Tabor, R.A., M.T. Celedonia, F. Mejia, R.M. Piaskowski, D.L. Low, B. Footen, and L. Park. 2004b. Predation of juvenile Chinook salmon by predatory fishes in three areas of the Lake Washington Basin.

Tabor, R.A., B.A. Footen, K.L. Fresh, M.T. Celedonia, F. Mejia, D.L. Low, and L. Park. 2007. Smallmouth bass and largemouth bass predation on juvenile Chinook salmon and other salmonids in the Lake Washington Basin. North American Journal of Fisheries Management 27: 1174–1188.

Tanner, C.M., F. Kamel, G.W. Ross, J.A. Hoppin, S.M. Goldman, M. Korell, C. Marras, G.S. Bhudhikanok, M. Kasten, A.R. Chade, and others. 2011. Rotenone, Paraquat, and Parkinson’s Disease. Environmental Health Perspective 119: 866–872.

[TID and MID] District, Turlock Irrigation, and Modesto Irrigation District. 2005. Ten year summary report pursuant to paragraph (G) of the 1996 FERC Order issued July 31, 1996. Report to Federal Energy Regulatory Commission for FERC Project 2299-024.

Tucker, M.E., C.M. Williams, and R.R. Johnson. 1998. Abundance, food habits and life history aspects of Sacramento squawfish and striped bass at the Red Bluff Diversion Complex, including the Research Pumping Plant, Sacramento River, California, 1994-1996. Red Bluff Research Pumping Plant Report Series, Volume 4. US Fish and Wildlife Service, Red Bluff, California.

Tyus, H.M. and J.F. Saunders III. 2000. Nonnative fish control and endangered fish recovery: Lessons from the Colorado River. Fisheries 25: 17–24.

Vigg, S., T.P. Poe, L.A. Predergast, and H.C. Hansel. 1991. Rates of consumption of juvenile salmonids and alternative prey fish by northern squawfish, walleyes, smallmouth bass, and channel catfish in John Day Reservoir, Columbia River. Transactions of the American Fisheries Society 120: 421–438.

Vogel, D.A., K.R. Marine, and J.G. Smith. 1988. Fish Passage Action Program for Red Bluff Diversion Dam: Final report of fisheries investigations. U.S. Fish and Wildlife Service Report No. FR1/FAO-88-19. 77 p.

Vredenburg, V.T. 2004. Reversing introduced species effects: Experimental removal of introduced fish leads to rapid recovery of a declining frog. Proceedings of the National Academy of Sciences of the United States of America 101: 7646–7650.

Wang, J.C.S. 2010. Fishes of the Sacramento-San Joaquin Estuary and adjacent waters, California: A guide to early life histories. Interagency Ecological Program Technical Report No. 9 (originally published 1986). Bureau of Reclamation, Mid-Pacific Region and Denver Technical Service Center.

Ward, D.L., J.H. Petersen, and J.J. Loch. 1995. Index of predation on juvenile salmonids by northern squawfish in the lower and middle Columbia River and in the lower Snake River. Transactions of the American Fisheries Society 124: 321–334.

Wu, B.J., and B.B. Bridges. In draft. Evaluating the use of carbon dioxide as an alternative predator removal technique to decrease Tracy Fish Collection Facility predator numbers and improve facility operations. Bureau of Reclamation, Tracy Series.

Zavaleta, E.S., R.J. Hobbs, and H.A. Mooney. 2001. Viewing invasive species removal in a whole-ecosystem context. Trends in Ecology & Evolution 16: 454-459.

Zimmerman, M.P., T.A. Friesen, D.L. Ward, and H.K. Takata. 2000. Development of a system-wide predator control program: Indexing and fisheries evaluation. Development of a System-Wide Predator Control Program: Stepwise Implementation of a Predator Index, Predator Control Fisheries, and Evaluation Plan in the Columbia River Basin. Annual report to the Bonneville Power Admin., Contract 94BI24514.