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FORUM

Projecting Cumulative Benefits of Multiple River Restoration Projects: An Example from the Sacramento-San Joaquin River System in California

G. Mathias Kondolf · Paul L. Angermeier · Kenneth Cummins · Thomas Dunne · Michael Healey · Wim Kimmerer · Peter B. Moyle · Dennis Murphy · Duncan Patten · Steve Railsback · Denise J. Reed · Robert Spies · Robert Twiss

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Abstract Despite increasingly large investments, the potential ecological effects of river restoration programs are still small compared to the degree of human alterations to physical and ecological function. Thus, it is rarely possible to "restore" pre-disturbance conditions; rather restoration programs (even large, well-funded ones) will nearly always involve multiple small projects, each of which can make some modest change to selected ecosystem processes and habitats. At present, such projects are typically selected based on their attributes as individual projects (e.g., consistency with programmatic goals of the funders, scientific soundness, and acceptance by local communities), and ease of implementation. Projects are rarely prioritized (at least explicitly) based on how they will cumulatively affect ecosystem function over coming decades. Such projections require an understanding of the form of the restoration response curve, or at least that we assume some plausible relations and estimate cumulative effects based thereon. Drawing on our experience with the CALFED Bay-Delta Ecosystem Restoration Program in California, we consider potential cumulative system-wide

G. M. Kondolf (✉)
Department of Landscape Architecture and Environmental Planning, University of California, Berkeley, 202 Wurster Hall #2000, Berkeley, CA 94720-2000, USA
e-mail: kondolf@berkeley.edu

P. L. Angermeier
United States Geological Survey, Virginia Cooperative Fish and Wildlife Research Unit, Virginia Polytechnic Institute and State University, Blacksburg, VA 24061-0321, USA

K. Cummins
California Cooperative Fish Research Unit, Humboldt State University, 1 Harpst Street, Wildlife & Fisheries Building, Room 212, Arcata, CA 95521, USA

T. Dunne
Bren School of Environmental Science and Management, University of California, Santa Barbara, Santa Barbara, CA 93106-5131, USA

M. Healey
Institute for Resources and Environment, University of British Columbia, Vancouver, BC, Canada V6T 1Z4

W. Kimmerer
Romberg Tiburon Center for Environmental Studies, San Francisco State University, Tiburon, CA 94920, USA

P. B. Moyle
Department of Wildlife, Fish and Conservation Biology, University of California, Davis, Davis, CA 95616, USA

D. Murphy
Department of Biology, University of Nevada, Reno, Reno, NV 89557, USA

D. Patten
Department of Land Resources and Environmental Sciences, Montana State University, Bozeman, MT 59717-3120, USA

S. Railsback
Lang, Railsback & Assoc., 250 California Ave., Arcata, CA 95521, USA
e-mail: Steve@LangRailsback.com

D. J. Reed
Department of Geology and Geophysics, University of New Orleans, New Orleans, LA 70148, USA

R. Spies
Applied Marine Sciences, P.O. Box 315, Little River, CA 95456, USA

R. Twiss
P.O. Box 422, Ross, CA 94957, USA

benefits of a restoration activity extensively implemented in the region: isolating/filling abandoned floodplain gravel pits captured by rivers to reduce predation of outmigrating juvenile salmon by exotic warmwater species inhabiting the pits. We present a simple spreadsheet model to show how different assumptions about gravel pit bathymetry and predator behavior would affect the cumulative benefits of multiple pit-filling and isolation projects, and how these insights could help managers prioritize which pits to fill.

Keywords River restoration · Chinook salmon · Sacramento River · San Joaquin River · Restoration response curves · Gravel augmentation

Introduction

Interest in and funding for river restoration is increasing, with over 37,000 restoration projects documented in the United States since 1990, and funding averaging more than \$1 billion annually (Bernhardt and others 2005), not including costs of several major ongoing restoration programs, such as those in the Florida Everglades, Chesapeake Bay, Columbia River Basin, and Colorado River in the Grand Canyon. Despite the magnitude of this investment, the scale of river restoration is still small compared to the scale of historical anthropogenic landscape change. If we expect to restore populations of fish and other organisms that either migrate through large ecosystems or otherwise depend on large-scale ecosystem connectivity for their survival, we need to be strategic about our restoration investments, and consider how many small projects may affect ecosystem function on the catchment scale.

Even large-scale river restoration programs are unlikely to return riverine ecosystems to pre-disturbance conditions or to an intended reference state in the foreseeable future. For example, the much-heralded artificial high-flow release from Glen Canyon Dam of 1275 $m^3 s^{-1}$ (45,000 cfs) in 1996 (Marzolf and others 1998) was less than half the pre-dam mean annual flood. The CALFED Bay-Delta Program, encompassing the San Francisco Estuary system and its watershed in northern California (Fig. 1), is one of the largest ongoing restoration programs in the nation, with more than \$500 million invested in restoration projects from 1997 to 2004 (CALFED 2005). Yet when we look at the results of these and other restoration efforts to date in the context of habitat losses and fish population declines since European settlement in 1850, it is clear that even a restoration effort on this scale will not reverse large-scale historical changes.

Restoration projects in CALFED region have been incremental, and generally small in relation to the historical losses. The area of tidal marsh in the San Francisco Bay was about 74,000 ha in 1850, and only about 6000 ha by 1990. These tidal marshes have proven among the easiest habitats to restore, largely because large areas of diked wetlands were held by relatively few landowners and the degradation (mostly conversion to salt ponds) could be undone mostly by restoring connection to tidal action. Collectively, tidal wetland restoration projects in the San Francisco Estuary have restored about 4400 ha (Jeff Haltiner personal communication May 2008), about 6.5% of the estimated 68,000 ha of tidal marsh lost in the San Francisco Estuary since 1850 (Bay Institute 1998). The ongoing South Bay Salt Ponds project will restore over 5000 ha, another 7.4% of the area historically lost. Looking upriver at riparian habitat in the Sacramento-San Joaquin River system, we see comparable historical decline, but less extensive restoration. Current riparian habitat in the San Joaquin River valley is about 7000 ha, about 5% of the 127,000 ha estimated to have been present in 1850 (Bay Institute 1998). We do not have figures for total areas of riparian habitat restored, but these will be much less than the tidal wetlands restoration because it is more difficult to restore large areas of riparian habitat because of altered river hydrology, fragmented land ownership, and extensive physical alterations to channels and floodplains.

Understanding such historical changes in the system is prerequisite to designing a realistic restoration strategy in the current, highly altered system. Conducting a historical analysis does not imply that one intends to recreate historical pre-disturbance conditions, but it helps us understand what changes may be irreversible and to judge whether restoration goals are realistic (Kondolf and Larson 1995).

We do not mean to imply that the incremental benefits of small habitat restoration projects cannot be effective, especially when even a small increase in available habitat or flow could increase survival rate or population persistence for an endangered species. We need not demand that successful restoration completely recover the system to its pre-disturbance level. It may be enough to turn the trend around, with individual projects seen as first steps in recovery. Nonetheless, it is useful to keep the historical and large-scale changes in mind, lest the success of having completed a successful restoration project lets us lose sight of the larger context. It took decades to cause the alterations, so it should not be surprising if restoration of highly altered river systems requires a long-term effort. The question becomes how to best allocate the (always limited) available resources strategically to achieve realistic restoration goals.

Restoration actions are typically reviewed and funded individually, based on factors such as consistency with programmatic goals of the funders, scientific soundness, and acceptance by local communities. However, if we intend to restore an ecosystem, we need to understand the

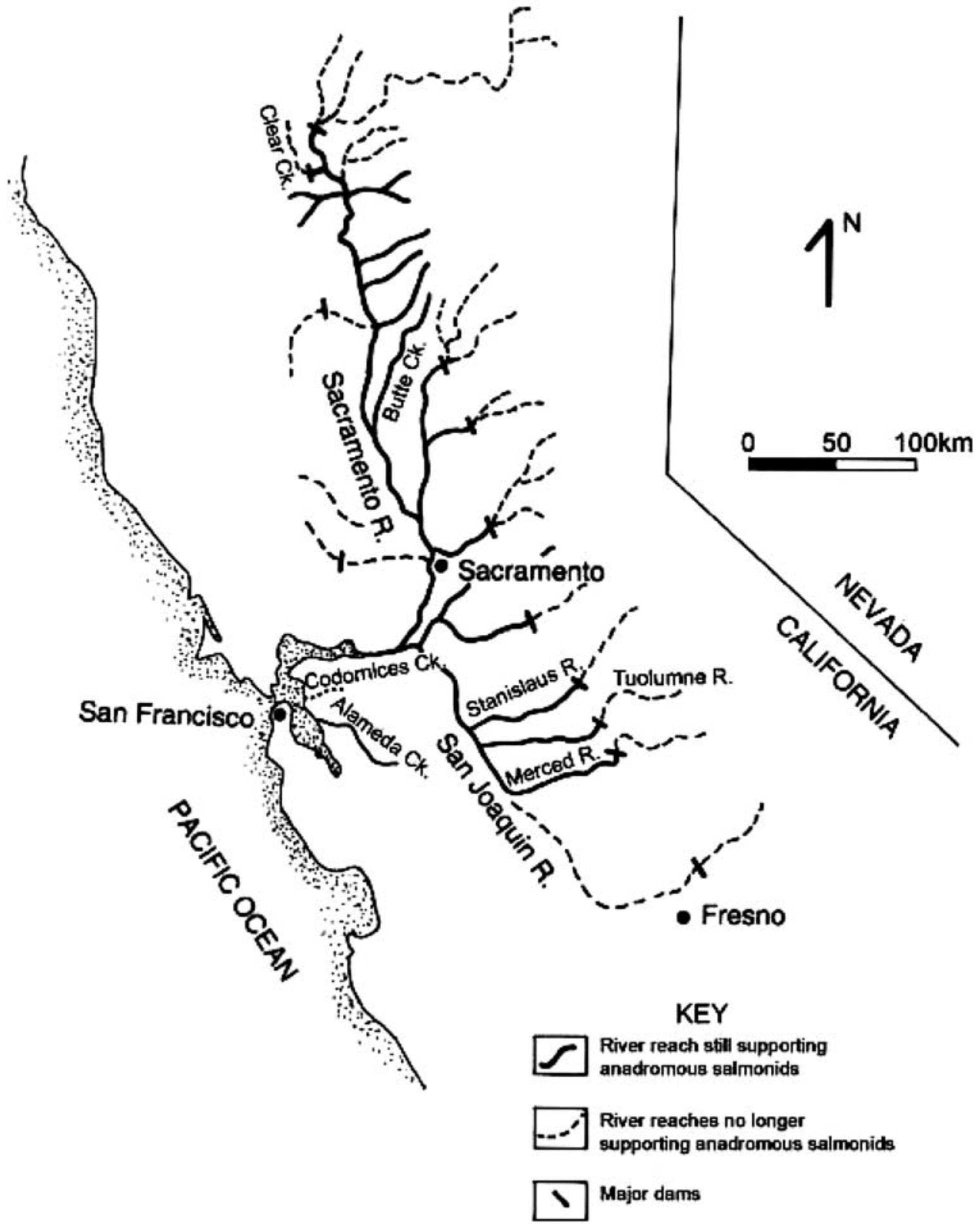

Fig. 1 Map of the San Francisco Estuary and its principal catchment area, the Sacramento-San Joaquin River system, showing major dams that block fish migration and interrupt water flow and sediment transport

cumulative effect of many small projects over large spatial and temporal scales. We must ask not only what kind of restoration actions to take, but also how to distribute those actions in time and space. Is it better to make small investments in many rivers or large investments in one or a few? Before we can answer this question, or even determine how to approach it, we need to articulate goals and objectives, based on a sound understanding of the physical and ecological processes and how they have changed over time. A long-term, basin-wide conceptual model can provide a framework for identifying restoration needs and evaluating the potential contributions of different projects. For a large river basin with multiple restoration objectives, this is not easy to do, especially as no single expert, nor even one discipline, can capture all the important factors.

The largest concentration of river restoration projects in North America has been along the Pacific coast, driven by efforts to increase populations of anadromous salmon (Bernhardt and others 2005). Many of the projects implemented in this region have involved restoration or enhancement of freshwater spawning habitat, in an effort to increase salmon populations. Decreases in spawning success caused by fine sediments (Kondolf 2000) may affect populations of salmon whose abundance is limited by spawning, but may have no effect on populations of salmon whose populations are limited by other life stages, such as juvenile survivorship in fresh water (Everest and others 1987). Thus, restoration projects that increase spawning success would be more likely to result in population increases where spawning habitat is limiting populations.

Furthermore, species at low population abundance may be limited by their biotic capacity to increase rather than the quantity of some limiting habitat. This example illustrates the utility of a limiting factor analysis, identifying critical points in the life history as a guide to where restoration funds should be invested. Targeting the factors that currently limit populations may not be a complete solution in itself, because relieving one limiting factor could fail to result in increasing salmon populations if another factor limits the population at a slightly higher level. Nonetheless, an understanding of the factors actually limiting fish populations is essential for strategic restoration planning.

Even for restoration projects that do address limiting factors, unless we clearly identify what specific changes in system dynamics are needed to achieve ecological goals, and then set out to effect that level of change, there is no a priori reason to expect that our restoration actions will “fix” the system. Current and proposed large-scale restoration programs in North America are commonly collections of large and small projects, each designed to improve some aspect of the system, but without a clear overarching framework (as provided by a conceptual model) to show how much of the necessary improvement each project will achieve, and whether the cumulative effects will be sufficient to achieve the large-scale goals. For example, a National Research Council review of a proposed US Army Corps of Engineers program to reduce losses of coastal wetlands in Louisiana concluded that most of the individual projects proposed “were scientifically sound, but taken together they do not represent the type of integrated, large-scale effort needed for such a massive undertaking” (NRC 2005).

Even in the absence of a comprehensive, system-wide model of degradation and restoration, an analysis of the cumulative effects of multiple small projects could help inform decisions about restoration investments. In this paper we suggest such projections of cumulative effects of multiple projects, and present an approach in which the incremental effects of many small projects are modeled over time to project their cumulative benefits in relation to system-wide restoration needs. We draw upon our experience serving as members of the CALFED Ecosystem Restoration Program (ERP) Science Board and modeling fish populations to consider some key issues related to prioritizing river restoration projects, the implications of different possible forms for restoration trajectories, and how the cumulative effects of multiple small projects might be projected into the future.

Prioritization of Projects

Based on our collective experience with the CALFED Bay Delta Ecosystem Restoration Program, where we looked at implemented restoration projects, we see that many have been relatively easy to implement and encountered little resistance, but don’t necessarily address the most critical ecosystem needs. Some have high monetary cost, but low political and administrative transaction costs. For example, the CALFED ERP has invested heavily in installation of fish screens, with \$31 million spent or allocated through 2001, with another \$55 million proposed in 2002. Though most diversions are privately owned, “for legal and historic reasons, most fish screens in California are paid for with public funds” (Moyle and Israel 2005:22). These projects appear to have been undertaken despite a paucity of scientific studies documenting the effectiveness of screens and without full analysis of many potentially negative side effects, such as fixing the river channel in place (Moyle and Israel 2005). However, there were few political objections to installing fish screens, and they were highly visible, “concrete” projects easily implemented.

In the early days of the CALFED ERP, there was not a systematic, comparative process for prioritizing projects; prioritization was often based on ease of implementation or intuitive appeal to agencies or stakeholders. Positive examples included removing barriers to reopen reaches of suitable habitat to anadromous salmon, as successfully accomplished on Butte and Clear creeks, Sacramento River tributaries (Fig. 1). There were good reasons for seeking projects that could be easily implemented. Politically it was important to have some visible, implemented projects to show results and to build momentum for the restoration program. Given this political context, it made sense to start on projects for which the barriers to implementation were easily surmountable. This has been the case for river restoration projects worldwide: Most rivers have been altered in multiple ways, but usually restoration projects and programs affect only some of these human alterations—the ones that can be feasibly restored given the political and economic context. As a result, the trajectories of restoration rarely parallel the preceding trajectories of degradation (Kondolf and others 2006). However, there is no rational reason to assume that easily-implemented projects will necessarily affect the factors limiting target species, substantively advance restoration goals, or otherwise rank among the highest ecological priorities.

A related problem occurs when restoration projects are not commensurate in spatial scale with the anthropogenic disturbance they are intended to counteract. Habitat restoration projects tend to be small “…because of logistical and resource constraints, and because ecological values must be balanced with the economic and social values derived from ongoing human pursuits” (Bond and Lake 2003:195). However, “…most degradation has occurred across large areas of the landscape, often whole catchments. Yet most efforts at habitat restoration are pitched as much smaller scales, typically individual sites or stream reaches" (Bond and Lake 2003:195). The habitat patches created by such small projects may be too small to support species with large foraging areas and home ranges, and thus the restorations are often simply not sufficient to cause measurable change to species persistence, biodiversity, or numbers of organisms. Another implication of the mismatch in scale between restoration efforts and historical anthropogenic change is that the effects of local habitat restoration projects may be easily overwhelmed by continuing broad-scale disturbances, such as drought, which are beyond the influence of the restoration project (e.g., Bond and Lake 2005).

As the role of science in the CALFED ERP has strengthened since 1997, better scientific justification has been required for projects, and better monitoring and evaluation have been implemented. Despite this progress, the program has not yet incorporated a system-wide model as a basis for setting priorities among competing projects.

Projecting Future Trends of Restoration

In evaluating the potential contribution of a restoration action, it is important to project its likely effect over time, in combination with other anticipated restoration projects, as well as external factors such as continued urbanization in contributing catchments, future water abstractions, and climate change. One of the most basic questions is what shape the degradation-restoration trajectory will take. Sarr (2002) proposed three different types of responses of range condition to overgrazing by livestock, and recovery of range condition following cessation of grazing (Fig. 2). These different degradation-recovery curves may be informative for restoration more broadly. In the elastic or rubber-band response, degradation from grazing is reversible along a subparallel trajectory to the degradation trajectory (Fig. 2a). In the Humpty-Dumpty response, changes are irreversible (Fig. 2b). In the broken-leg response, degradation is reversible, but the recovery trajectory is very different and may involve a long lag time before ecological effects of reduced stress are manifest (Fig. 2c).

The rubber-band response might occur along streams where grazing pressure was not severe and channel form and watershed soils are intact (Sarr 2002). Applying this concept to aquatic restoration, we could imagine a system in which salmonid populations remained robust basin-wide, but access to habitat in one tributary has been blocked by a dam. Removal of the dam might result in rapid recolonization of the upstream reach by strays from other tributaries in the system (an elastic response). However, the system may have undergone changes such that simply re-connecting the reach is insufficient to restore the

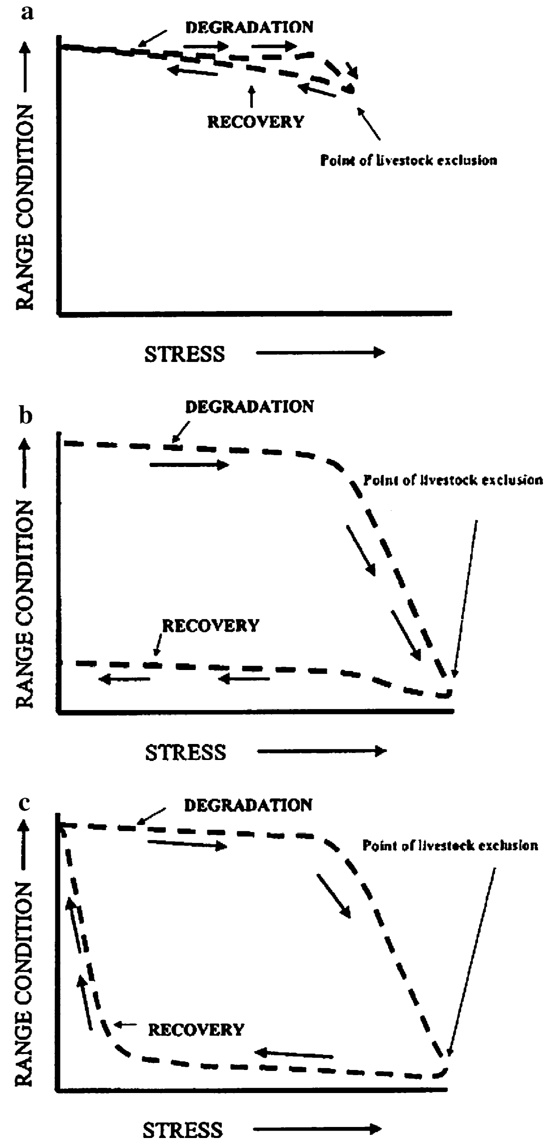

Fig. 2 Trajectories of degradation and restoration, developed by Saar (2002) to model response of riparian vegetation to removal of grazing pressure. (**a**) the elastic or rubber-band model, in which degradation is reversible along a trajectory roughly parallel to the degradation; (**b**) the Humpty-Dumpty model, in which changes are irreversible; and (**c**) the broken-leg model, in which degradation is reversible, but with substantial hysteresis in the trajectory and possibly a significant lag in response

ecosystem function (irreversible degradation): the formerly dammed tributary may have undergone morphological changes that have degraded the habitat, or invasive species may have become established, displacing natives from critical habitats. Population levels throughout the system may have been severely reduced by other stressors, and there may not be enough fish in the system to repopulate the newly-opened habitat, at least for some years. With removal of stressors and successful reproduction, population levels may eventually increase, although with

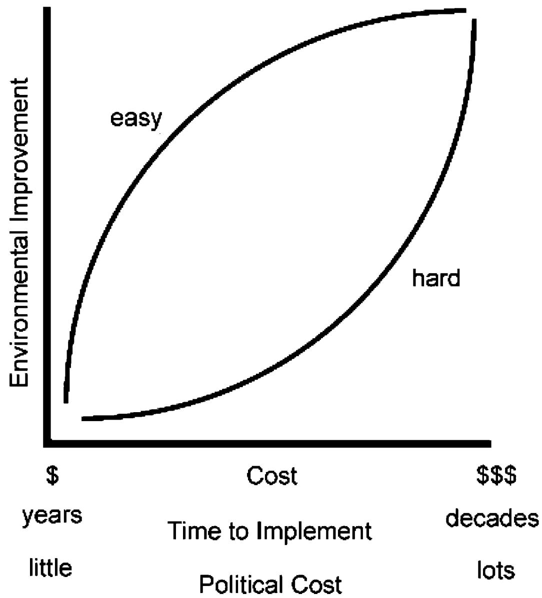

Fig. 3 Trajectories of investment and the resulting restoration. "Easy" projects are those for which a small investment promptly yields an environmental improvement, while "hard" projects are those for which considerable investment (financial and political) is needed over a long time before significant improvement is manifest. (redrawn from Schmidt 2005)

hysteresis resulting in a delayed or reduced response to early restoration actions. Similar questions may be posed for restoration of invertebrate communities in streams, where recolonization is affected by such factors as proximity of the restored reach to pools of potential colonists, barriers to colonization, and introduced species (Bond and Lake 2003).

We can imagine different potential restoration response curves depending on the underlying physical and ecological processes and interactions. Schmidt (2005) proposed different trajectories of investment expended and resulting restoration (Fig. 3). He used the term "easy" for projects where a small investment promptly yields an environmental improvement, and "hard" for projects where considerable investment (financial and political) is needed over a long time before significant improvement is manifest. If the expected response curves can be specified for different restoration actions, it could help decision makers prioritize projects, especially in light of public expectations of success. In an example cited by Schmidt (2005), river restoration efforts on the Colorado River below Glen Canyon Dam are very expensive and may not yield visible improvements until a very large investment is made, while smaller investments in other reaches of the river may yield ecological benefits more quickly.

One problem facing managers is that we usually don't know the shape of the restoration curve a priori, so initially we must do projections based on current understanding including the degree of uncertainty. Indeed, restoration trajectories may be highly complex and rarely possible to fully observe, suggesting that theoretical models may serve a useful role in projecting possible (and testable) outcomes (Anand and Desrochers 2004).

An Example from the Sacramento-San Joaquin River System: Isolating and Repairing Disused Gravel Pits to Reduce Juvenile Salmon Predation

Gravel Pits Along the Merced and Tuolumne Rivers

Commercial mining of sand and gravel for construction aggregate (e.g., for concrete) has been extensive along the San Joaquin River and its major tributaries, the Merced, Tuolumne, and Stanislaus rivers. Much of the early mining extracted aggregate directly from the active channel bed, and as this source was exhausted, from floodplain pits. In many cases, these floodplain pits were adjacent to the active channel, separated by only a narrow strip of unexcavated land. Many of these pits have been "captured" by the river such that the river now flows through the aggregate pits (Fig. 4a) (Kondolf 1998). The captured pits create a number of environmental problems. First, they are longitudinal discontinuities in the river. The river abruptly changes from a flowing water body (lotic) to a still-water pond (lentic). The pits trap sand and coarser sediments carried downstream by the river, interrupting the longitudinal continuity of sediment transport and starving the downstream reach of sediment. The bed slope at the upstream end of the captured pit is over-steepened, and regressive erosion (the upstream migration of a headcut) typically ensues. Thus, gravel mining—either in the channel or in captured gravel pits—typically results in channel incision.

The lentic environments created by the captured gravel pits did not exist naturally in these rivers, and native fish are not adapted to them. However, the pits provide excellent habitat for exotic warmwater fish, notably largemouth bass (*Micropterus salmoides*) and smallmouth bass (*M. dolomieu*), which are voracious predators of small fish such as juvenile salmon (Moyle 2002). Largemouth bass were first introduced in California in 1891, and were quickly spread throughout the state by anglers and agency biologists (Moyle 2002). They usually inhabit "warm, shallow…waters of moderate clarity and beds of aquatic plants…in low elevation streams above the Central Valley. They occur mostly in disturbed areas where there are large, permanent pools with heavy growths of aquatic plants…" (Moyle 2002:398–399). They can tolerate a wide range of temperatures and poor water quality, including dissolved oxygen concentrations of only 1 mg/liter (Moyle 2002:399).

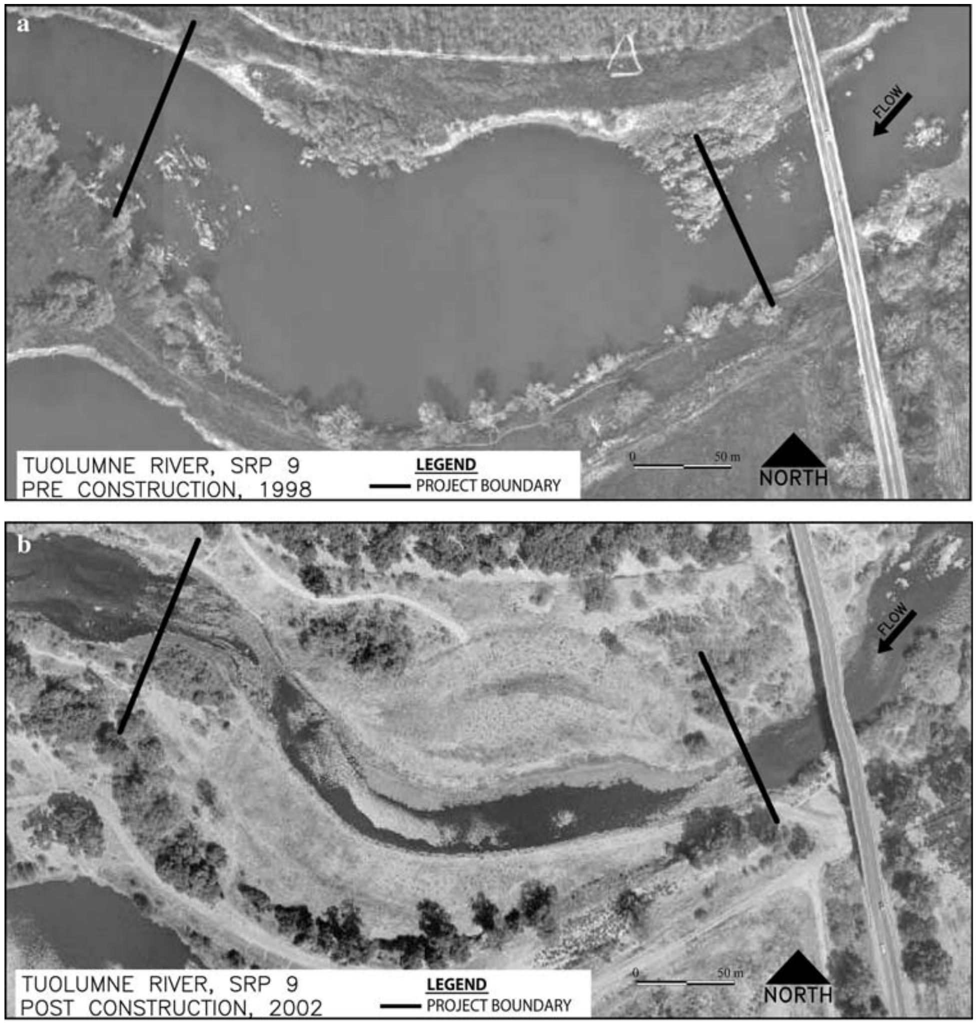

Fig. 4 Aerial photographs of the Tuolumne River about 66 km upstream of the San Joaquin River confluence, showing the former gravel pit designated locally as SRP 9 in (**a**) 1998, prior to restoration; and (**b**) in 2002, after restoration. (Photos courtesy of Scott McBain)

The California Department of Fish and Game (1987, as cited in EA 1992) indicated that large- and smallmouth bass stomachs contained young salmon and estimated predation rates by largemouth bass of 0.6 salmon per day in 1989, and between 1.1 and 1.6 salmon per day in 1990; two thirds of salmon smolts on the Tuolumne River were estimated to be lost to predation (EA 1992). The original report is not widely available and apparently has not been peer reviewed, but this information seems consistent with what is known about the habits of both bass and salmon.

Despite the lack of solid data on predation rates and their impact on juvenile salmon populations, predation of juvenile salmon by exotic bass is widely viewed as a significant limiting factor for salmon in tributaries of the San Joaquin River, and captured gravel pits are believed to provide important habitats to support the predatory bass populations. Thus, in an effort to reduce predation on outmigrating juvenile salmonids, a number of restoration projects have isolated gravel pits from the channels of the Merced and Tuolumne rivers, reconstructing a confined channel more typical of the pre-disturbance state of these rivers (Fig. 4). To date, reaches affected by gravel mining have been restored in three projects on the Tuolumne River and two projects on the Merced, affecting over 11 km of river at a total cost of $29 million (not including agency staff time and related costs) (Table 1). The projects ranged

Table 1 Gravel-mining channel restoration projects completed to date on the Merced and Tuloumne Rivers

River/project	Length of river (km)	Total cost ($ millions)
Tuolomne River		
SRP 9	0.8	2.6
7/11 Reach	4.2	7.2
Ruddy	1.9	6.5
Merced River		
Ratzlaff	0.8	4.8
Robinson	3.7	8.2

Total to date: $29.3 million

Robinson project includes 115 ha floodplain habitat easement

Source: Rhonda Reed, CALFED ERP, unpublished data 2003 widely in area affected, amount of material moved, and consequently cost, but averaged just under $6 million each, with funding from CALFED, US Fish and Wildlife Service, and the California Departments of Fish and Game and Water Resources. Although these projects are often thought of as pit-filling and isolation projects, some have included other actions to permit restoration of fluvial processes through the restored reaches, such as setting back mining levees that had confined the low-flow channel. Although the main purpose of the restoration was to reduce predator (bass) habitat, these projects were also designed eventually to restore other riverine functions such as limited dynamic channel migration, continuity of sediment transport, and establishment of a riparian forest.

Despite the considerable costs, the pit repair/channel restoration projects to date have fixed only a small percentage of all the pits along the Tuolumne and Merced. An inventory of captured pits on the Merced showed some sixteen pits remaining after completion of the Ratzlaff and Robinson restoration projects (Table 2; Stillwater Sciences 2002). An inventory on the Tuolumne showed eight gravel pits and "special-run pools," unnaturally deep reaches that were evidently former in-channel gravel mining sites, of which three have been restored to date by pit repair/channel reconstruction projects: special-run pool (SRP) 9, the 7/11 segment, and the M.J. Ruddy segment (Table 3; McBain and Trush 2000). McBain and Trush (2000) estimated that repair costs for special-run pools 5, 6, and 10 would total about $8.5 million.

Thus, the task of repairing captured gravel pits has been only partly accomplished to date, raising the question of what would be the net effect of these completed projects if further funding were cut off tomorrow? Put another way, what is the shape of the response curve? Of the forms proposed by Sarr (2002) (Fig. 2), which might apply here? Some important factors influencing the response curve could include the geometry and bathymetry of captured pits, which may influence predator spawning, abundance, and feeding efficiency. The remaining, unrepaired pits have not been systematically surveyed to determine their shape, especially the distribution of water depths. However, it could be expected that some pits are more amenable to supporting bass populations. While bass occur in waters up to 6 m in depth, they are more abundant in depths of 3 m or less, and for spawning they require "sand, gravel, or debris-littered bottoms at depths of 0.5–2 m" (Moyle 2002:400). Thus, adult bass can live in deep waters but may not reproduce there, favoring shallows with rooted macrophytes. Shallows are also likely more productive of prey that bass rely on during seasons when juvenile salmon are absent (e.g., small fish and large invertebrates). This suggests that pits with adjacent extensive shallow water

Table 2 Inventory of instream and captured floodplain gravel pits along the Merced River

San Joaquin River (distance above confluence, km)	Pit ID	Mine name	Length (m)	Width (m)	Depth (m)	Volume required (m^3)	Volume required (tons)
100.1–101.2	GM1-C1	unknown	457.2	243.84	No data	68,000	74,000
90.4–91.1	GM1-C2	unknown	304.8	60.96	2 to 3	13,000	14,000
87.2–88.3	GM1-C3	unknown	670.56	121.92	1.5 to 2.5	43,600	48,000
93.5–95.0	GM1-C4	unknown	609.6	30.48	1.2	3000	3000
Not reported	GM1-T1	Carson Pit I	853.44	701.04	No data	No data	No data
Not reported	GM1-T2	Carson Pit II	335.28	137.16	No data	No data	No data
108	GM1-T3	Silva Expansion	No data	No data	No data	No data	No data
85.4–87.9	GM1-T4	Bettencourt Ran	1371.6	457.2	No data	No data	No data
81.1–82.7	GM2-C1	River Rock	609.6	182.88	1.2 to 4	93,300	102,000
77.2–80.5	GM2-C2	Silva/Turlock Ro	1005.84	121.92	4 to 6	243,000	265,000
73.9–74.3	GM2-C3	Turlock Rock	426.72	60.96	7	79,500	87,000
70.0–70.5	GM2-C4	Cressey Sand and Gravel	426.72	91.44	3.4 to 8.9	101,000	110,000
68.7–69.8	GM2-C5	Turlock Rock	548.64	243.84	3	143,000	156,000
80.1–80.8	GM2-T1	Turlock Rock	640.08	274.32	No data	No data	No data
76.4–76.9	GM2-T2	Turlock Rock	243.84	182.88	6	115,000	126,000
75.1–75.9	GM2-T3	Turlock Rock	487.68	152.4	6	No data	No data

GM = gravel mining reach; C = in channel pit; T = Terrace pit. The designation "C" in the Pit ID indicates a "captured" pit; "T" indicates a noncaptured pit. Captured pits are those for which restoration should be considered, but noncaptured pits are also listed, as they may become captured in the future

Source: Stillwater Sciences, Merced River Corridor Restoration Plan, 2002. No cost estimated information provided, but report notes that restoration is costly, and references a range of $4.5 to $7.5 milion based on projects undertaken on other reaches along this river (Ratzlaff Reach [km 64–65]; Robinson Reach [km 67–70])

Table 3 Gravel-mining impacted reaches for restoration, Tuolumne River

Special run pool projects					
River km (distance above San Joaquin River confluence, km)	Pit designation	Width (m)	Depth (m)	Volume required (m^3)	Cost to repair ($)
84.7–85.9	SRP 5	170[a]	NR	135,000	1,463,000
77.7–79.5	SRP 6	150[a]	NR	170,000[b]	2,334,000
66.0–66.6	SRP 9—already restored	120	2–6	110,000[c]	2,700,000[d]
64.9–66.0	SRP 10	120	11	225,000	4,657,000

Gravel mining reach restoration				
River (km)	Segment designation	Width (m)	Amount of Imported Material Required (m^3)	Cost to repair ($)
97.0–104	7/11 segment—already restored	NR	320,000	NR
91.7–97	MJRuddy segment—already restored	NR	355,000	NR
90.6–94.3	Warner/Deardorff segment	NR	On-site Source	NR
87.7–90.6	Reed segment	NR	On-site Source	NR

NR = not reported; SRP = special run pool

[a] Floodway width at this point; no separate width for pit given

[b] Summation of two amounts listed as needed: 60,000 yd^3 + 159,000 yd^3

[c] Additional fill of 3500 yd^3 will be required to fill 4 backwaters connected to SRP 9 and SRP 10

[d] Includes cost of restoring the connection between SRP 9 and SRP 10

Source: McBain and Trush (2000), US Fish and Wildlife Service, Anadromous Fish Restoration Program, Habitat Restoration Plan for Tuolumne River Corridor

areas, likely to support bass reproduction, might be reasonable targets for isolation and filling.

For the purposes of this discussion we assume that predation on migrating salmon by bass in the gravel pits is a significant source of mortality, whose reduction would substantially enhance the survival of the salmon. We would expect this assumption to be rigorously tested as part of a program to determine the optimum approach to restoring the river, but despite the expenditures discussed, this has not been done. If predation is important, an obvious question is the extent to which predation would decline with each successive pit repair. If 30% of the pit volume or area were filled or isolated, would we expect predation to decline by 30%? Or would the relation between total pit volume and bass population be nonlinear, due to effects such as differences in pit geometry or migration among pits by the predators?

Incremental Analysis of Isolating Captured Gravel Pits

How might we expect smolt survival downstream to increase as gravel pits were progressively filled? To illustrate a range of possible responses as functions of assumed ecological relationships, we developed a set of spreadsheet models, assuming 10 gravel pits in series, through which smolts must migrate. We looked at how the overall smolt survival (number left alive after going past all pits divided by the number that start) varies with the number of pits that are filled in, assuming that filling in a gravel pit completely eliminates predation risk. All models assume that predator (bass) populations are independent of smolt abundance (i.e., that bass populations are controlled by factors other than the consumption of smolt prey), and we do not consider year-to-year variation in either predator or smolt populations, only the fate of smolts in one year during outmigration.

First, we assume that all 10 pits are of equal size and shape, pits have the same number of predators in each, and that predators do not move among pits. We assume that each predator eats a fixed number of smolts: this assumption is met if predation is limited by the number of smolts a predator can digest during the time that smolts are moving through (a “Holling Type II functional response” with prey saturation; Holling 1959). (Other details of the model, such as parameters for number of smolts eaten per predator, do not affect the outcome we are interested in: the shape of the response of smolt survival to number of pits restored.) In this case, smolt survival increases linearly with the number of pits filled in (Model 1, Fig. 5a). (For Fig. 5a, we assumed that a cohort of 100,000 smolts starts through a series of *n* pits, and in each pit there are 400 bass that each

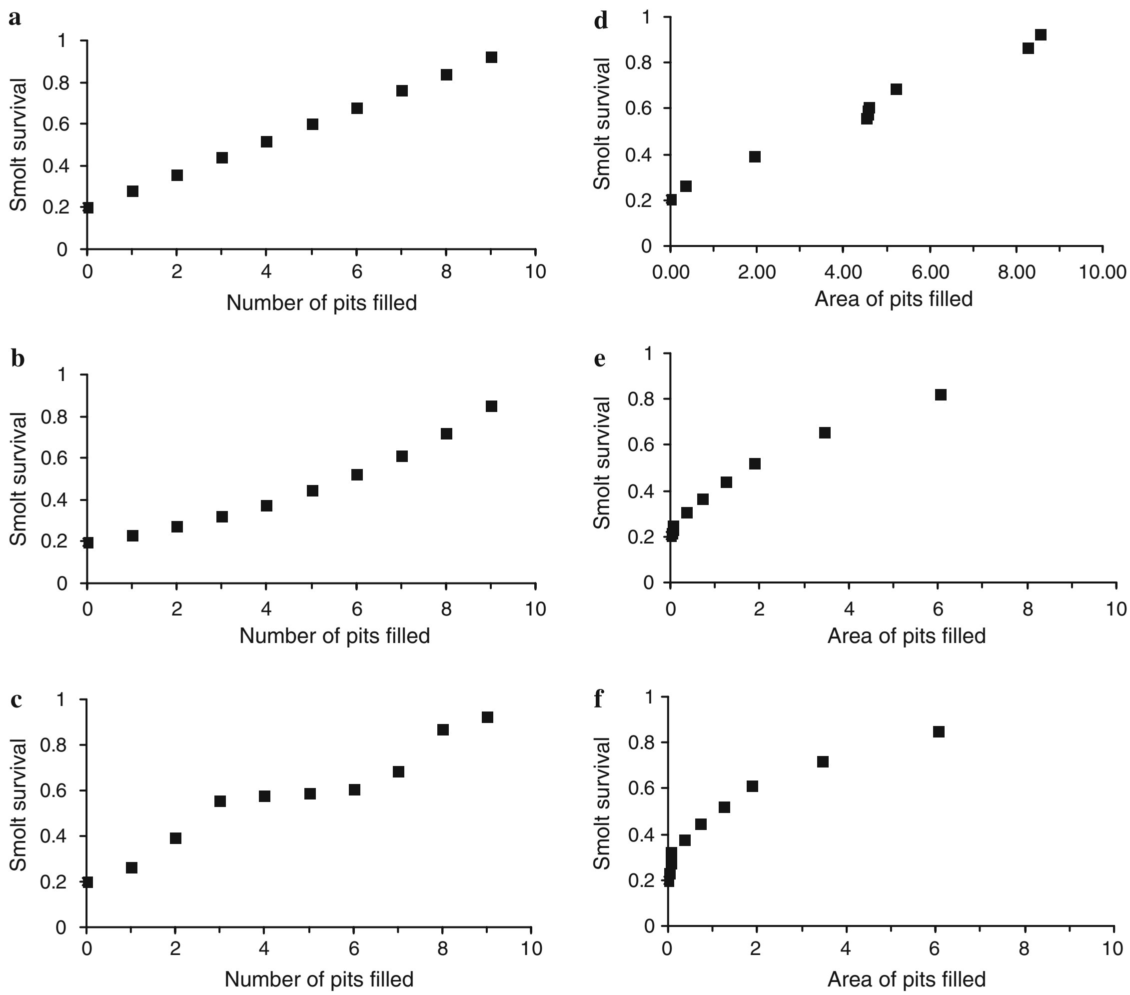

Fig. 5 Spreadsheet models of overall smolt survival through a series of 10 gravel pits as a function of the number of pits filled/isolated to eliminate predation risk. All models assume that predator (bass) populations are independent of smolt abundance (i.e., that bass populations are controlled by factors other than the consumption of smolt prey), and do not consider year-to-year variation in either predator or smolt populations; only the fate of smolts in one year during outmigration. (**a**) Model 1, assuming all 10 pits are of equal size and shape, each have the same number of predators, and predators do not move among pits. We assume that each predator eats a fixed number of smolts (limited by the predator's capacity to digest during the period of smolt migration). (**b**) Model 2, assuming that predators move among pits in a way that optimizes (and equalizes, since all predators are the same) predator intake, as predicted by the "ideal free distribution" theory. (**c**) Model 3, like Model 1 except that the pits vary in size. We assume that the number of predators per pit varies with its shoreline circumference, assigned each pit a radius drawn from a uniform random distribution, and assumed that pits are filled in randomly, without considering their size. (**d**) Model 4, same as Model 3 except we show how smolt survival varies with the area of pit filled in (as a surrogate for the cost of filling them). (**e**) Model 5, still assuming bass populations are related to shoreline circumference, but filling pits from the smallest (with the highest ratio of shoreline circumference, and therefore predators, to area) to the largest. (**f**) Model 6, assuming an ideal free distribution (as in Model 2) and showing smolt survival benefit per cost, filling pits from smallest to largest

eat a total of 20 smolts. Hence, survival (S) through the sequence of pits is: $S = \frac{100{,}000 - 8000n}{100{,}000}$).

Next, we assume that predators move among pits in a way that optimizes (and equalizes, since all predators are the same) predator intake, as predicted by the "ideal free distribution" theory (Fretwell and Lucas 1970). This results in predators being densest in the first pit and least dense in the last pit, because the density of smolts decreases as they move through the series of pits. The consequence of such an ideal free distribution is that the mortality risk is the same in each pit. In this case, survival increases exponentially with the number of pits filled in: filling in the first few pits has little benefit, but the benefit per pit increases as more are filled in (Model 2, Fig. 5b). (In Fig. 5b, we assumed that bass redistribution is such that they eat 15% of smolts passing through each pit, which matches the mortality over 10 pits of Model 1. Here: $S = 0.85^n$).

Now, let's assume the conditions in the first model, except that the pits vary in size. We assume that the number of predators per pit varies with its shoreline circumference, for the reasons discussed above for why shallow water is important to bass populations. We assigned each pit a radius drawn from a uniform random distribution, and assume that pits are filled in randomly, without considering their size. In this case, survival increases with the number of pits filled in, but in an unpredictable way (Model 3, Fig. 5c). (The 10 pits used in Fig. 5c were assumed circular with randomly drawn shoreline lengths of 2.6, 1.9, 6.2, 2.8, 0.5, 0.5, 0.6, 5.7, 4.5, and 2.1 arbitrary units. We assumed 146 bass, that each eats 20 smolts, per unit of shoreline length; these parameters reproduce the 10-pit mortality of Model 1. Hence:

$$S = \frac{100,000 - \sum_{i=1}^{n} 2920L_i}{100,000}$$

where L_i is the shoreline length of pit i).

The modeled benefit of filling each pit depends on its size. However, when we look at how smolt survival varies with the area of pit filled in (as a surrogate for the cost of filling them), then the response is fairly linear (Model 4, Fig. 5d).

We now continue to assume this relation of bass population to shoreline circumference, but fill pits from the smallest (which has the highest ratio of shoreline circumference, and therefore predators, to area) to the largest. In this case, the smolt survival benefits per cost (as area filled in) decreases as more pits are filled (Model 5, Fig. 5e). Similarly, if we assume an ideal free distribution (so, as in Model 2, 15% of smolts entering each pit are assumed eaten in it) and plot the smolt survival benefit per cost, we find the benefits decrease rapidly as more pits are filled in, when pits are filled from smallest to largest (Fig. 5f).

Under the assumptions of these last two models, especially that bass populations vary with shoreline length, more value per restoration dollar is obtained by filling in the smaller (or higher-shoreline) pits first. In the absence of data on actual pit geometries, we assumed that all pits have similar shoreline shape and depth, so area of shallow habitat used by bass is proportional to shoreline length, and pit volume is proportional to area. This result is contrary to intuition, which might suggest that the biggest problem (i.e. pit) should be fixed first. However, if pit bathymetry is available, we could more defensively prioritize shallow pits for filling first.

This analysis ignores how the number of smolts eaten one year could affect the abundance of either bass or smolts in future years. Such effects seem likely and would create even more complex responses than illustrated here. We also don't consider other strategies to reduce predation, such as increasing fishing pressure, which has been shown to reduce bass populations in reservoirs (Moyle 2002). We present these models not to argue that any one is necessarily an accurate representation of restoration response, but to illustrate the potential for planning to improve the cost-effectiveness of multiple restoration actions.

Conclusions

The San Francisco Estuary system illustrates the degree to which the magnitude of even a large restoration program can be dwarfed by the magnitude of historical anthropogenic change. Understanding the extent of these historical changes helps us put current restoration efforts into perspective, and helps us see that in most cases it is impossible to restore "pre-disturbance conditions." These insights also suggest that the long-term incremental effects of many small restoration projects need to be projected to help decision-makers prioritize allocation of current restoration funds. The CALFED Ecosystem Restoration Program, from which we have drawn the examples in this article, has made tremendous progress incorporating science and scientific review in its decision-making process, but long-range projections of the cumulative effects of multiple small projects have not been attempted.

Looking at one restoration activity, we can see how such analysis might help the planning process. The use of gravel to fill and restore disused gravel pits along the Merced and Tuolumne rivers has several key uncertainties that call into question the value of filling a small number of pits while leaving others unfilled. Furthermore, the cost of filling all pits will be very high. This situation illustrates the value of scientific knowledge; in this case better information on the degree of predation, the likely response of the predation rate to changes in river form and fishing pressure, and better bathymetric data could enable managers to defensibly prioritize pits for isolation and filling, improving the response function. Alternatively, better knowledge of the broader, ecosystem effects of restoring dynamic channel processes would help to determine whether predator control would suffice or whether the value of a restored river merited the full restoration program. We suggest that an analytical approach to choosing reasonably among various plausible models is needed to help managers proceed cost-effectively. Are there some experiments and/or monitoring that need to be done to choose wisely? Can adequate rankings of the realism of various models be developed from expert opinion? In any case, this example illustrates the importance of drawing up explicit conceptual models to inform planning decisions.

If further pit repair and channel reconstruction projects are implemented we suggest they be designed with long-term river-wide restoration potential in mind. One element that might be considered in restoring a future dynamic channel is a more natural flow regime, with deliberate high flow releases to move sediment, etc. Thus, pit repair/channel reconstructions would not be designed only for the current, much-reduced flow regime, but would be capable of passing potentially higher future flows. Additionally, an incremental-analysis approach could be applied at a regional spatial scale to examine how salmon populations would respond to extensive channel repairs in a few versus many versus all rivers in the Sacramento-San Joaquin River system.

Even with generous, continuous funding, the CALFED ERP and allied restoration programs simply do not have the resources to restore the entire Sacramento-San Joaquin River system to its pre-disturbance condition. This is also the case in other river systems with large restoration programs, such as the Mississippi Delta, the Colorado, and Columbia rivers. We submit that these restoration programs are better viewed as collections of many small projects, and it is only by targeting restoration actions to specific areas that the available restoration funds will be adequate to produce a tangible result. Political and economic realities constrain restoration programs such that they typically affect only some structures and functions of the pre-disturbance system (Kondolf and others 2006). To realistically assess what can be achieved with existing resources, not only for charismatic species like salmon but for other ecosystem functions, will require an interdisciplinary scientific approach that includes projections of the long-term cumulative benefits of many small projects. While the US generally lacks powerful basin-scale authorities and a system for assessing restoration needs on a system-wide scale (Baron and others 2002), the European Union's Water Framework Directive is putting into place such "competent authorities" for basin-scale analysis and management, providing a framework for prioritizing the types and locations of restoration projects (Scheuer 2005). Applied to rivers in the US, such an approach could provide a framework for more strategic investments in river restoration (Grantham and others 2008).

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References

Anand M, Desrochers RE (2004) Quantification of restoration success using complex systems concepts and models. Restoration Ecology 12(1):117–123

Baron JS, Poff NL, Angermeier PL, Dahm CN, Gleick PH, Hairston NG Jr, Jackson RB, Johnston CA, Richter BD, Steinman AD (2002) Meeting ecological and societal needs for freshwater. Ecological Applications 12:1247–1260

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

Bay Institute (2003) The Bay Institute ecological scorecard, San Francisco Bay index. The Bay Institute of San Francisco, San Rafael, California

Bernhardt ES, Palmer MA, Allan JD, Alexander G, Barnes K, Brooks S, Carr J, Clayton S, Dahm C, Follsted-Shah J, Galat D, Gloss S, Goodwin P, Hart D, Hassett B, Jenkinson R, Katz S, Kondolf GM, Lake PS, Lave R, Meyer JL, O'Donnell TK, Pagano L, Powell B, Sudduth E (2005) Sythesizing U.S. river restoration efforts. Science 308:636–637

Bond NR, Lake PS (2003) Local habitat restoration in streams: constraints on the effectiveness of restoration for stream biota. Ecological Management and Restoration 4(3):193–198

Bond N, Lake PS (2005) Ecological restoration and large-scale ecological disturbances: effects of drought on a stream habitat restoration experiment. Restoration Ecology 13:39–48

CALFED Bay-Delta Program (2000) Strategic plan for ecosystem restoration. Sacramento

CALFED Bay-Delta Program (2005) Ecosystem restoration multi-year program plan (years 6–9) and annotated budget (year 5). DRAFT. June 16, 2005

California Department of Fish and Game (1987) The status of San Joaquin drainage Chinook salmon stocks, habitat conditions and natural production factors. Prepared for the State Water Resources Control Board Bay/Delta hearing process, exhibit 15

EA (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)

Everest FH, Beschta RL, Scrivener JC, Koski KV, Sedell JR, Cederholm CJ (1987) Fine sediment and salmonid production: a paradox. In: Salo EO, Cundy TW (eds) Streamside management: forestry and fishery interactions. University of Washington, Institute of Forest Resources, Contribution No. 57. pp 98–142

Fretwell SD, Lucas HL (1970) On territorial behavior and other factors influencing habitat distribution in birds. Acta Biotheoretica 19:1–39

Grantham T, Christian-Smith J, Kondolf GM, Scheuer S (2008) A fresh perspective for managing water in California: insights from applying the European water framework directive to the Russian River. Available on line: http://www.lib.berkeley.edu/WRCA/WRC/pubs_contri.html#208

Holling CS (1959) Some characteristics of simple types of predation and parasitism. Canadian Entomologist 91:385–398

Kondolf GM (1998) Environmental effects of aggregate extraction from river channels and floodplains. In: Bobrowsky P (ed) Aggregate resources: a global perspective. Balkema, Rotterdam, pp 113–129

Kondolf GM (2000) Assessing salmonid spawning gravels. Transactions of the American Fisheries Society 129:262–281

Kondolf GM, Larson M (1995) Historical channel analysis and its application to riparian and aquatic habitat restoration. Aquatic Conservation. 5:109–126

Kondolf GM, Boulton A, O'Daniel S, Poole G, Rahel F, Stanley E, Wohl E, Bang A, Carlstrom J, Cristoni C, Huber H, Koljonen S, Louhi P, Nakamura K (2006) Process-based ecological river restoration: visualising three-dimensional connectivity and dynamic vectors to recover lost linkages. Ecology and Society 11 (2):5. [online] URL: http://www.ecologyandsociety.org/vol11/iss2/art5/

Marzolf GR, Valdez RA, Schmidt JC, Webb RH (1998) Perspectives on river restoration in the Grand Canyon. Bulletin of the Ecological Society of America 1998:250–254

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

Moyle PB (2002) Inland fishes of California. University of California Press, Berkeley

Moyle PB, Israel JA (2005) Untested assumptions: effectiveness of screening diversions for conservation of fish populations. Fisheries 30(5):20–28

NRC (National Research Council) (2005) Drawing Louisiana's new map: addressing land loss in coastal Louisiana. National Academy Press, Washington

Sarr DA (2002) Riparian livestock exclosure research in the western United States: a critique and some recommendations. Environmental Management 30(4):516–526

Scheuer S (2005) EU environmental policy handbook—A critical analysis of EU environmental legislation, published by the European Environmental Bureau, September 2005. Available online at http://www.eeb.org/publication/policy_handbook.htm

Schmidt JC (2005) A general approach for establishing national priorities for river restoration, presentation to the American Geophysical Union Spring Meeting, May 2005, New Orleans

Stillwater Sciences (2002) Merced River corridor restoration plan. Stillwater Sciences, Berkeley, California