Fisheries

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Collapse of San Francisco Bay Pelagic Fishes

Streambank Restoration Effectiveness

Using an Individual-Based Model to Estimate Multiple Watershed Effects

A Responsible Approach

Juvenile mullet can be tagged with Coded Wire Tags (above) which can then be recovered from adults (right) to evaluate hatchery programs.

A half century ago, efforts to supplement marine fish stocks in the United States were abandoned for lack of evidence of their success. Since then, worldwide declines in coastal fisheries have sparked a resurgence in hatchery-based marine stock enhancement. New aquaculture and tagging technologies, along with demands for accountability in fisheries management, have resulted in a quantitative approach to marine stock enhancement[1].

For example, Dr. Ken Leber at Mote Marine Laboratory (www.mote.org) in Florida conducts research that addresses critical uncertainties about stock enhancement of important coastal commercial and recreational species. In a recent publication[2], Dr. Leber used Coded Wire Tags to estimate the postrelease mortality of striped mullet *Mugil cephalus* released at different sizes. He found that size-dependent postrelease mortality had a significant impact on the cost-effectiveness of stocking strategies. Dr. Leber has also used Coded Wire Tags in his research evaluating the effectiveness of stocking snook, Pacific threadfin, and red snapper.

This research, and many other programs examining marine stock enhancement around the world rely on Coded Wire Tags to identify and track hatchery reared fish and crustaceans after release. Please contact us if we can help with your program.

[1]Blankenship H. L. and K. M. Leber, 1995. A responsible approach to marine stock enhancement. American Fisheries Society Symposium 15: 167-175.
[2]Leber, K. M., R. N. Cantrell and P. Leung. 2005. Optimizing cost-effectiveness of size at release in stock enhancement programs. North American Journal of Fisheries Management. 25:1596–1608.

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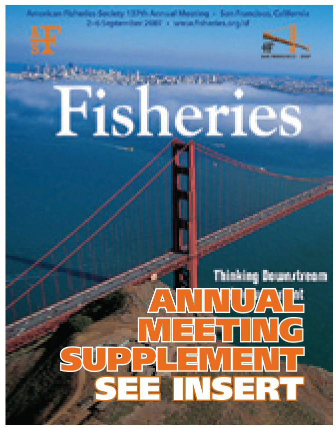
DEPARTMENTS



COVER: "Thinking Downstream and Downcurrent," 2007 AFS Annual Meeting poster art.
CREDIT: Design by Rad Smith/Garcia and Associates and art by Rene Reyes.

The American Fisheries Society (AFS), founded in 1870, is the oldest and largest professional society representing fisheries scientists. The AFS promotes scientific research and enlightened management of aquatic resources for optimum use and enjoyment by the public. It also encourages comprehensive education of fisheries scientists and continuing on-the-job training.

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COLUMN: PRESIDENT'S HOOK

Jennifer L. Nielsen

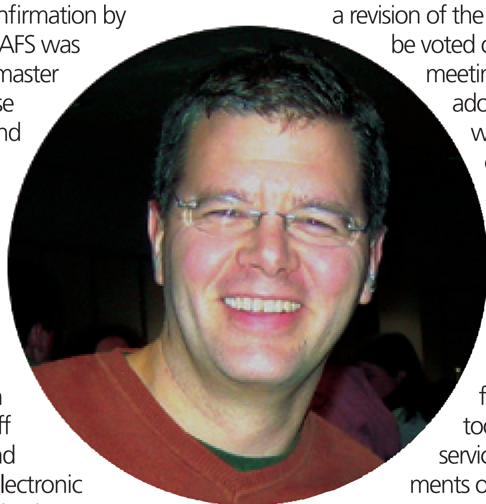
AFS President Nielsen can be contacted at jlnielsen@usgs.gov.

Member-centric Information Technology at AFS

A group of invited members and information technology (IT) leaders met with AFS Executive Director Gus Rassam and the AFS staff in Bethesda on 16–17 May 2007 to discuss AFS IT problems, issues, and electronic services. Over several years we have had significant problems with information technology at AFS, culminating with a web site crash without back up just before the Annual Meeting in Lake Placid. Up to this point, IT services at AFS have been developed and organized as a one-way street with society-to-membership functions available on the AFS web site. Some tools such as individual Unit membership databases were only available through requests to the Bethesda staff, with results that were often outdated and far from adequate for Unit needs. Electronic services were limited and often outsourced by the Bethesda office, or lacked integration among the AFS office and the various Units or membership leaders. This IT summit was proposed at the Governing Board mid-year meeting in Atlanta and was an effort to focus on the causes of these problems and short-term and long-term corrections available to AFS, as well as a chance to chart a course for future IT opportunities within AFS. Our meeting in Bethesda was very productive and many issues were addressed that are important to the general membership.

Recently many changes have been made to the AFS IT structure in Bethesda. Two new servers are now in place and dedicated to AFS IT with daily back up and security updates. Soon two T1 lines will facilitate faster and more efficient electronic communications at the Bethesda office. New computers with updated software have been put in place for most of the AFS staff. We now have two dedicated AFS IT staff and the main office is working on opening access to the Units for many general service activities such as electronic registration, abstract submissions, and online voting services. These are significant improvements, but much more will happen given the short-term (i.e., before the Annual Meeting in San Francisco) and long-term strategies developed in our Bethesda meeting for AFS IT.

First and foremost it was determined that IT within AFS needs to become more member-centric, with a strong focus on how information technology and transfer can facilitate our membership and Unit needs. Participants suggested we develop a bi-level web site where AFS will have an open-access web site available to the general public for posting general information on AFS business, policy, education, and outreach that we want to highlight to the public. The second level would be a members-only, user-friendly web site where interactive access would be through electronic membership identification and confirmation by individual members. AFS was tasked to develop a master membership database that is coordinated and updated instantly through any activity at this members-only web location. These data will be tied to an active membership list, organized and quality controlled by both membership and staff on a regular basis, and linked to other AFS electronic services such as publications and meeting services. This structure will improve our electronic directory database on membership status, dues, and contact information. This level of web access will also contain members-only activities, notifications, bulletin boards, membership information transfer opportunities, list serv access information, and potential "communities" for AFS Units that allow the exchange of documents, moderated discussions, and other information exchange among Unit members. We discussed new membership electronic services that may become possible soon through AFS, such as online audio streaming of Annual Meeting plenary talks and select symposia, AFS blogs, and new open-access publications for members only.



Many of the issues membership has had with AFS IT revolve around access and electronic tools needed by our Units for different reasons at different times. Updated Unit membership databases are needed, as are easy-to-use, effective electronic meeting registration and abstract submission software. Such electronic services should be developed and positioned by AFS for Unit use and easy membership access. As president of AFS, I proposed changing the name of the "Web Editorial Advisory Board" to the "Electronic Services Advisory Board" (ESAB). This name change requires a constitutional change and a revision of the board's mandate will be voted on at our business meeting in San Francisco. If adopted, the new ESAB will continue to provide oversight on the content, structure, and architecture of the AFS web site, but the board will also oversee other electronic services at AFS with a goal of focusing on electronic tools and user-friendly services as the IT requirements of our membership grow. Last October, I appointed **Joel Carlin** as chair of the ad hoc Electronic Services Committee that will shepherd this transition. Joel was a major contributor to the recent meeting in Bethesda. We tasked Joel and his committee with organization of an IT workshop at the San Francisco meeting that will be open to all Unit webmasters. This workshop is intended to provide a forum on the current state of IT development at AFS and to bring coordination among Units on needs and opportunities for electronic services in the future. I hope this IT workshop becomes a traditional activity at our Annual Meeting.

Continued on page 304

NEWS: FISHERIES

2006 STATUS OF U.S. FISHERIES REPORT

The National Oceanic and Atmospheric Administration (NOAA) recently released its annual report on the status of U.S. marine fisheries for 2006, tracking both population levels and harvest rates for species caught in federal marine waters between 3 and 200 miles off U.S. coasts. This report characterizes stocks under two categories: (1) subject to overfishing and (2) overfished. A stock that is subject to overfishing has a harvest rate above the level that provides for the maximum sustainable yield, while a stock that is overfished has a biomass level below a biological threshold specified in its fishery management plan.

In 2006, the report shows population levels for 187 fish stocks and multi-species groupings known as complexes, out of a total of 530 managed U.S. fisheries. Of these, 47 (25%) were overfished. NOAA scientists also assessed harvest rates for 242 stocks and found that 48 (20%) were subject to overfishing.

Two stocks were taken off the overfishing list, Gulf of Mexico vermilion snapper and Atlantic sea scallop; Gulf of Mexico vermilion snapper was also taken off the overfished list. However, six stocks were added to the overfishing list in this annual report (Gulf of Mexico gag grouper, Gulf of Mexico gray triggerfish, Atlantic dusky shark, Eastern Pacific yellowfin tuna, petrale sole, and winter skate) and six stocks were added to the overfished list (northern and southern stocks of monkfish, South Atlantic pink shrimp, Atlantic sandbar shark, porbeagle, and dusky shark).

Within one year of being notified that overfishing is occurring in a particular fishery, the responsible Regional Fishery Management Council must take action to address that overfishing. The stocks that have been identified as overfished will now require rebuilding plans. The recent reauthorization of the Magnuson-Stevens Fishery Conservation and Management Act contained a new mandate to end all overfishing by 2010 through the use of annual catch limits. All stocks, including those at sustainable levels, are required to have annual catch limits by 2011 in order to prevent future overfishing.

The report is available online at www.nmfs.noaa.gov/sfa/statusoffisheries/SOSmain.htm.

MORE MITTEN CRABS FOUND ON EAST COAST

GREGORY RUIZ, SMITHSONIAN ENVIRONMENTAL RESEARCH CENTER

Chinese mitten crabs, first reported in the Chesapeake Bay last year, also now have been caught in Delaware Bay during May 2007. In total, seven adult male mitten crabs have been documented from the two bays. The mitten crab is native to eastern Asia and has already invaded Europe and the western United States, where it has established reproductive populations. A Mitten Crab Network has been established to examine the abundance, distribution, and reproductive status of crabs in Chesapeake Bay, Delaware Bay and other estuaries along the eastern United States. The initial partnership between the Smithsonian, Maryland Department of Natural Resources, U.S. Fish and Wildlife Service, NOAA, and Delaware Division of Fish and Wildlife is now being expanded to include resource managers, commercial fishermen, research organizations, and citizens along the East Coast.

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FLASH
TECHNOLOGY

Synchronous Rise and Fall of Cod Stocks Points to Environmental Factors in Decline

A study of decades of population estimates for the various cod stocks off Canada and New England shows that the stocks grew and declined at about the same time, revealing that environmental factors played a stronger role than previously thought in the collapse of the cod fishery. In an article in the latest *Transactions of the American Fisheries Society,* Brian Rothschild of the University of Massachusetts Dartmouth points to a strong negative environmental signal, possibly associated with plankton dynamics, as a leading suspect in the cod's disappearance.

Since colonial times, cod has been the mainstay of New England and Canadian Maritime fishing fleets. The collapse of cod fisheries in the northwest Atlantic in the early 1990s hit both the industry and fishing towns hard, but the cod population still hasn't recovered despite radically reduced fishing. The cod's decline has been an intriguing scientific mystery filled with dozens of sometimes apparently conflicting clues. First came the cod population declines of the 1970s. This was followed by a strong upswing and then the steep decline beginning in the mid-1980s when fishing pressure was still low. Another clue was that cod were not only waning in numbers but were experiencing significant decreases in growth rate. Meanwhile, scientists painstakingly worked to tease apart the various stocks of cod, only to find that the extent to which the stocks mingle is still its own mystery and has unknown effects on the population as a whole. Finally, changing water temperatures seemed to be associated with salinity decreases and changing cod diets.

Rothschild assembles the various pieces of this puzzle into a coherent picture to answer the question of what happened to the cod. Since the abundance of various stocks from southern Newfoundland all the way to the Gulf of Maine rose and fell at the same time, complexes of cod stocks must have been responding to environmental factors operating over a wide area. The dramatically reduced slower growth of cod and their changing stomach contents support the concept that the supply of plankton may have been disrupted, hence affecting the availability of cod forage like capelin and herring that feed on plankton.

"These environmental changes were probably as important in influencing declines in cod abundance as the effects of fishing," said Rothschild. "The standing stock biomass and weight-at-age statistics for various stocks tend to follow the same pattern. However, when fishing is superimposed on top of an unfavorable environment, it appears to accelerate the negative effects of the environment in bringing about a decline."

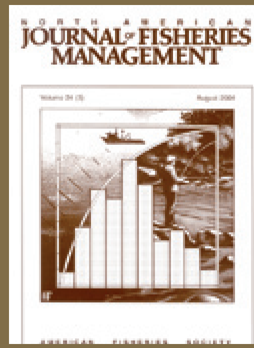
Rothschild noted these observations have important implications for fishery management. All of the "rules" used in fishery management: production, yield-per-recruit, and stock-and-recruitment relate to the effects of fishing and ignore the effects of the environment. The known strong influence of the environment on stock abundance suggests reevaluating definitions and remedies for overfishing. In particular, it needs to be recognized that rebuilding stocks in a mandated finite period of time may not be feasible. These observations are also critically important to the fishing industry. The industry needs to know whether decreases or increases in stock abundance are the result of fishing or environmental change. Causes associated with fishing suggest modifying the intensity of fishing, but causes associated with multi-annual environmental variability suggest longer-term strategies that might involve changing target species or investment strategies. Of greatest concern to the industry are questions related to longer-term changes. For example, are the observations on cod populations over the last several decades the harbinger of permanent changes in the ocean ecosystem that result from climate change signals in the North Atlantic Ocean?

"I think the most important point is that a decline in the cod populations was inevitable, and fishing simply aggravated it," said noted fisheries biologist Ray Hilborn, the Richard C. and Lois M. Worthington Professor of Fisheries Management at the University of Washington. "Fishing pressure should have been reduced sooner on the Canadian stocks, but they were going to decline regardless. The decline of the Gulf of St. Lawrence stocks in particular began at a time of high abundance and low fishing pressure."

Rothschild explained that avenues for future research focus upon developing capabilities to separate the influence of fishing from the influence of the ocean environment on fish stock variability. This capability must relate in a significant way to the as-yet-unresolved problem of understanding the variability in recruitment. What would be new in recruitment research, according to the author, would be the development of an observation system that could statistically resolve events in the ocean on scales relevant to a larval fish: hours and meters.

Coherence of Cod Stock Dynamics in the Northwest Atlantic by Brian J. Rothschild of the School for Marine Science and Technology, University of Massachusetts Dartmouth, New Bedford, Massachusetts. *Transactions of the American Fisheries Society* 136:858-874. Rothschild can be contacted at 508/910-6382 or brothschild@umassd.edu.

JOURNAL HIGHLIGHTS: North American Journal of Fisheries Management



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The Collapse of Pelagic Fishes in the Upper San Francisco Estuary

El Colapso de los Peces Pelágicos en La Cabecera Del Estuario San Francisco

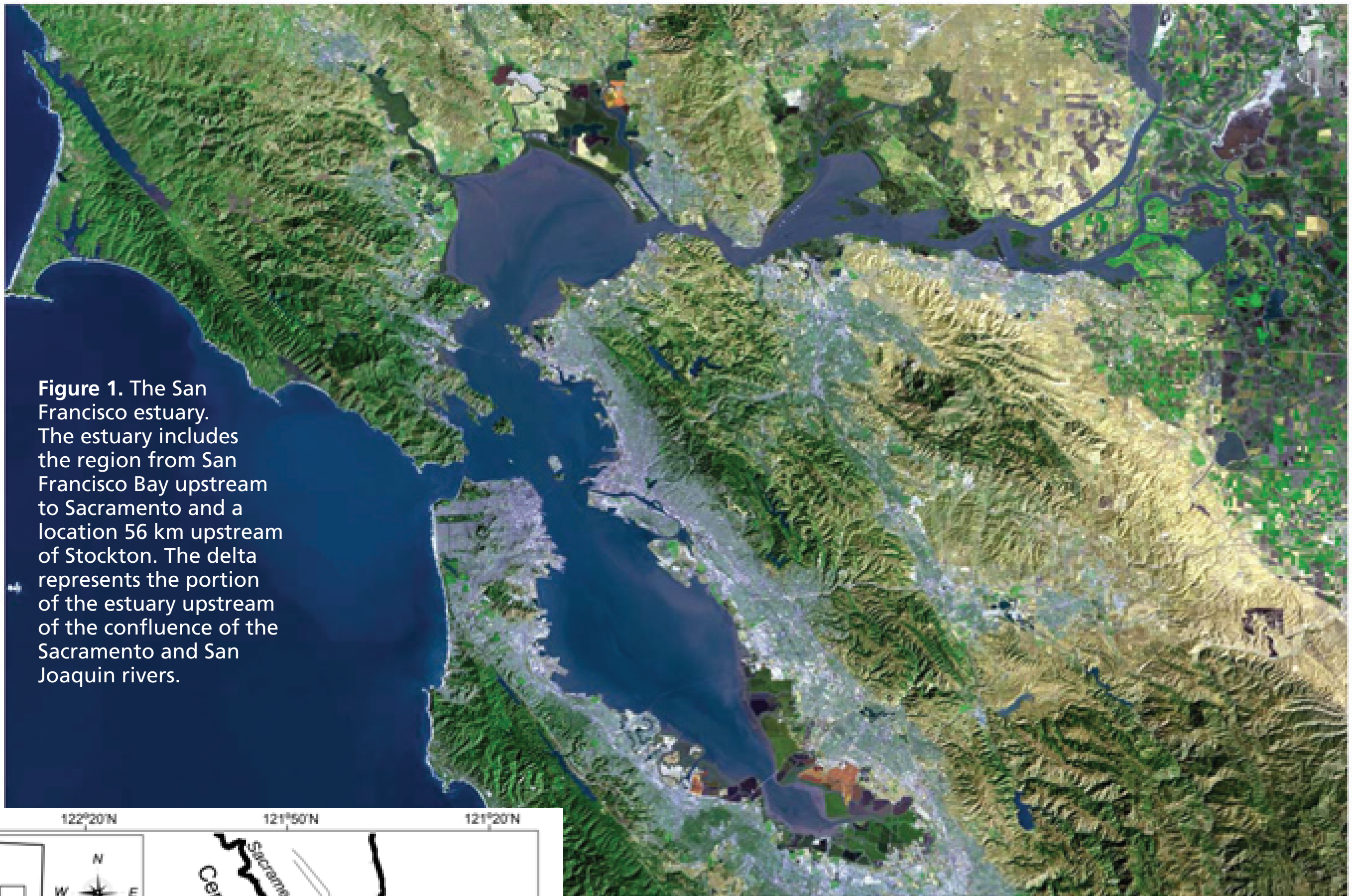
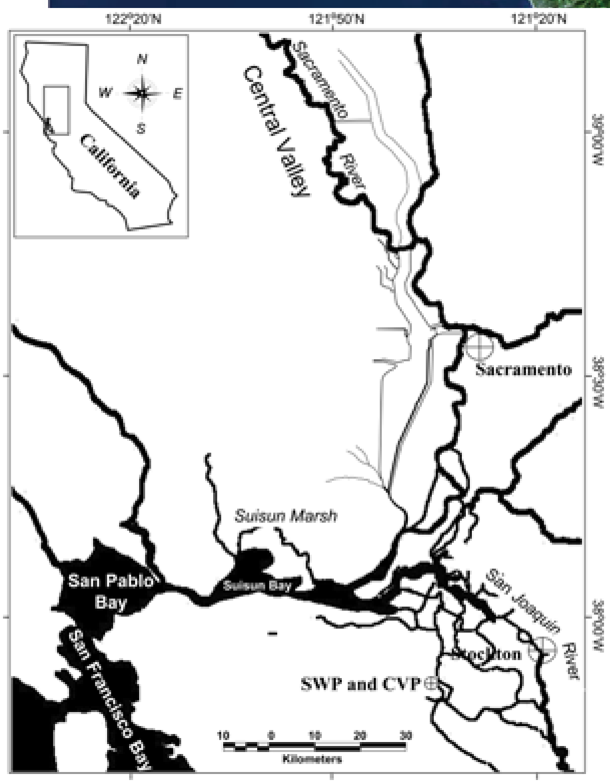


Figure 1. The San Francisco estuary. The estuary includes the region from San Francisco Bay upstream to Sacramento and a location 56 km upstream of Stockton. The delta represents the portion of the estuary upstream of the confluence of the Sacramento and San Joaquin rivers.



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ABSTRACT: Although the pelagic fish community of the upper San Francisco Estuary historically has shown substantial variability, a recent collapse has captured the attention of resource managers, scientists, legislators, and the general public. The ecological and management consequences of the decline are most serious for delta smelt (*Hypomesus transpacificus*), a threatened species whose narrow range overlaps with large water diversions that supply water to over 25 million people. The decline occurred despite recent moderate hydrology, which typically results in at least modest recruitment, and investments of hundreds of millions of dollars in habitat restoration and environmental water allocations to support native fishes. In response to the pelagic fish collapse, an ambitious multi-agency research team has been working since 2005 to evaluate the causes of the decline, which likely include a combination of factors: stock-recruitment effects, a decline in habitat quality, increased mortality rates, and reduced food availability due to invasive species.

RESUMEN: A pesar de que la comunidad de peces pelágicos de la cabecera del Estuario San Francisco históricamente ha mostrado una considerable variabilidad, su reciente colapso ha llamado la atención de manejadores, científicos, legisladores y público en general. Las consecuencias ecológicas y de manejo de dicha caída son particularmente graves para el "delta smelt" (*Hypomesus transpacificus*); una especie amenazada cuyo estrecho rango de distribución coincide con un gran reservorio hidrológico que suple de agua a más de 25 millones de personas. El colapso tuvo lugar a pesar de la modesta dinámica hidrológica del lugar, que al menos dio como resultado un reclutamiento igualmente moderado, y de una inversión de cientos de millones de dólares para la restauración del hábitat y el aseguramiento de cuerpos de agua que sirven de hábitat a los peces nativos. Como respuesta a la caída de los peces pelágicos, un ambicioso equipo de trabajo constituido por diversas agencias, ha venido trabajando desde 2005 para evaluar las causas del colapso, las cuales seguramente involucran diversos factores, tales como: efectos sobre la relación parentela-progenie, disminución de la calidad del hábitat, aumento en las tasas de mortalidad y una reducción en la disponibilidad de alimento debido a la presencia de especies introducidas.

Globally, the collapse of many of the world's fisheries remains the most important issue facing fisheries managers. The collapses are most pronounced in coastal regions, where declines have occurred on the scale of decades to millennia (Worm et al. 2006). With the 2007 American Fisheries Society Annual Meeting in San Francisco, the fisheries community will come together in close proximity to one of the more recent resource collapses in North America, the decline of pelagic fishes in the upper San Francisco Estuary (Figure 1). As in many other estuaries, the origin of this collapse dates back many decades, and coincides with increasing anthropogenic pressure (Lotze et al. 2006). However, an apparent recent change toward exceptionally low abundance indices for pelagic fishes caused great concern among California's resource managers, who had invested hundreds of millions of dollars in habitat restoration and environmental water for the upper San Francisco Estuary over the past decade. Our objectives in this paper are to describe the extent of the problem, its management consequences, and the evolving research effort to identify the causes.

THE SAN FRANCISCO ESTUARY

The San Francisco Estuary is the largest estuary on the U.S. Pacific Coast (Figure 1). It is formed by the confluence of two major sources of water: ocean water transported into the estuary by tides and freshwater runoff from small Coast Range streams and the Sacramento-San Joaquin watershed, which drains 40% of California's surface area including the western slope of the Sierra Nevada and the Central Valley. The estuary grades from marine dominance in central and southern San Francisco bays to freshwater dominance in the Sacramento-San Joaquin Delta. Suisun and San Pablo bays are the regions of greatest salinity variation, which occurs primarily through mixing of seawater with freshwater inflow from the delta. The northern part of Suisun Bay is fringed by Suisun Marsh, the largest contiguous wetland along the Pacific coast of the western United States.

The estuary has been heavily modified since California's Gold Rush in the mid-nineteenth century (Atwater et al. 1979; Nichols et al. 1986). A timeline of some of the major alterations is provided as Table 1, reflecting the long-term habitat modifications, frequent species introductions, and changes to hydrology. Over the past 150 years, large-scale reclamation of marshes fringing south San Francisco Bay, Suisun Marsh, and the delta for agriculture, mu-

Table 1. Timeline of some of the major anthropogenic changes to the San Francisco estuary.

Event	Year(s) of Occurrence
Hydraulic gold mining	1849-1884 [a]
Channelization and wetland reclamation	1860-1930[a]
Early fish introductions	1871-1908[b]
Contra Costa Canal Diversion	1940
Shasta Dam closure	1942
Friant Dam closure	1948
Central Valley Project Tracy Pumping Plant	1951
Threadfin shad introduction	1954-1963[c]
Oroville Dam construction and closure	1957-1968
State Water Project Banks Pumping Plant	1963-1969[d]
Clifton Court Forebay	1974
Overbite clam introduction	1986
Period of rapid nonnative copepod invasions	1963-1994[e]
Bay-Delta Accord signed	1994[f]

[a] Mount (1995)
[b] This was the period of most intentional sport fish introductions including striped bass (*Morone saxatilis*), American shad (*Alosa sapidissima*), carp (*Cyprinus carpio*), and several species of centrarchidae and ictaluridae.
[c] Threadfin shad (*Dorosoma petenense*) was introduced into southern California in 1954. It was detected in upper San Francisco Estuary fishery surveys by 1963 (Turner 1966).
[d] Increasing numbers of pumps came online during this period.
[e] Increasing shipping traffic and associated ballast water releases during this period led to the establishment of seven zooplankton species in the upper San Francisco Estuary: *Oithona davisae, Limnoithona sinensis, Sinocalanus doerri, Pseudodiaptomus forbesi, Tortanus dextrilobatus, Acartiella sinensis,* and *Limnoithona tetraspina* (Kimmerer and Orsi 1996; Kimmerer 2004)
[f] The Bay Delta Accord resulted in substantial changes delta outflow and export requirements (Koehler 1995)

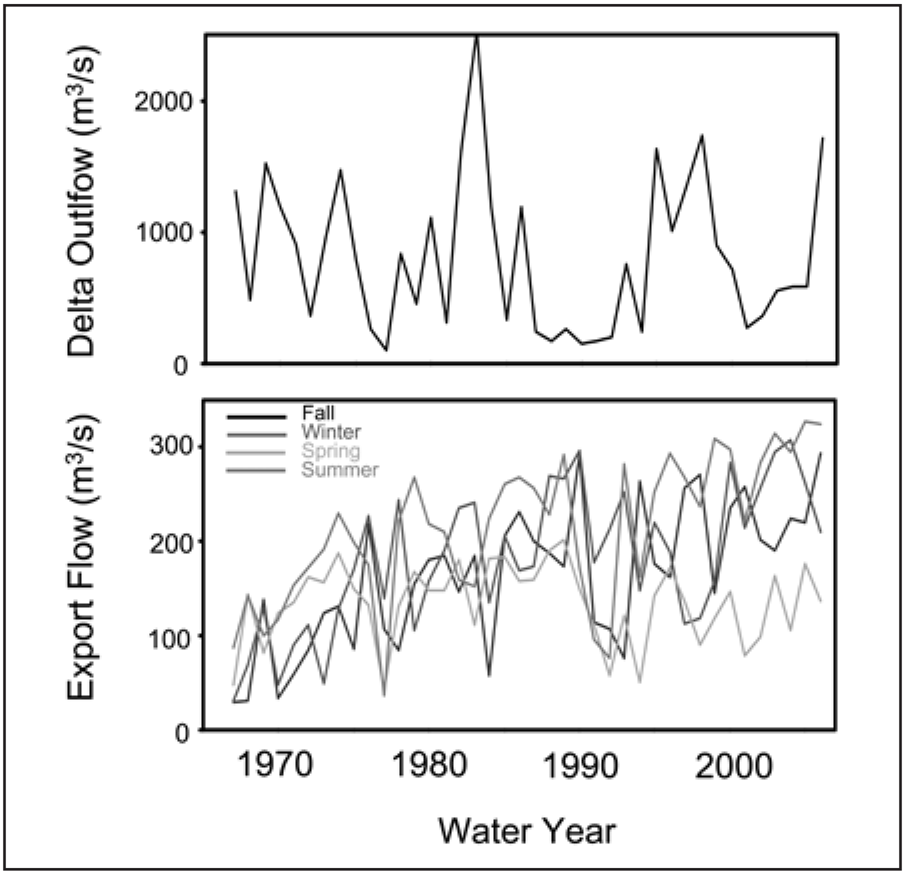
nicipal, and industrial uses removed 95% of historical wetlands from the estuary. Other principal changes included channelization and dredging of rivers, removal of large woody debris, substantial alteration of the flow regime, and introduction of numerous exotic organisms. As an indication of the degree of alteration by introduced species, the estuary has been called the most invaded on the planet (Cohen and Carlton 1998). Additional changes are likely in the near future: for example, the quagga mussel (*Dreissena bugensis*) was discovered in southern California in late 2006. In the likely event that the quagga mussel invades the upper San Francisco estuary, it could have effects similar to zebra mussels (*Dreissena polymorpha*), a close relative that has severely degraded other regions of the United States (Strayer et al. 1999).

Table 2. Summary of annual export volumes (km^3) from the four state and federal water diversions in the Sacramento-San Joaquin Delta for water years following the Bay-Delta Accord (1995-2005). The Contra Costa and Tracy diversion facilities are part of the federal Central Valley Project (CVP). The Harvey O. Banks and North Bay Aqueduct diversion facilities are part of the State Water Project (SWP).

Water diversion	1st year of operation	Average volume (range)
Contra Costa	1940	0.15 (0.12–0.23)
Tracy (CVP)	1951	3.10 (2.60–3.50)
Banks (SWP)	1968	3.60 (2.10–4.90)
North Bay Aqueduct	1988	0.05 (0.03–0.07)

During the past 60 years, the delta has been increasingly maintained as a permanent freshwater environment through large-scale regulation and manipulation of river flows in order to maintain high quality water for agriculture, municipal, and industrial uses. Two large water diversions and two smaller diversions in the delta (Figure 1), components of the State Water Project (SWP) and federal Central Valley Project (CVP), are allowed to export up to 35%-65% of river inflows depending on the time of year (Table 2; Figure 2). More than 2,500 smaller, privately-owned water diversions are also scattered throughout the Suisun Bay/Marsh and delta to supply water for municipalities, waterfowl management, and agriculture (Herren and Kawasaki 2001). The combined net annual diversion rate from these smaller facilities is 2 km^3, comprising a substantial fraction of water use in the delta (Kimmerer 2002a).

Figure 2. Delta outflow (m^3/s) averaged over water years (top) and export flow (m^3/s) averaged over seasons (bottom). Water years begin on 1 October of the previous calendar year. Seasons are in 3-month increments starting in October. Export flows are the sum of diversions to the federal Central Valley Project and State Water Project pumping plants. The outflow and export data are from California Department of Water Resources (http://iep.water.ca.gov/dayflow).



The fish community of the San Francisco Estuary is especially rich (e.g., Matern et al. 2002; Feyrer and Healey 2003; Nobriga et al. 2005), with 87 species collected since 1993 from just two of the sampling programs—the fall midwater trawl conducted by the California Department of Fish and Game (DFG) and salvage of fishes at the screens of the SWP pumping plant (http://baydelta.water.ca.gov/). Species richness is inflated by the presence of introduced species, which comprised over 40% of the total number reported in the two surveys. As in other estuaries (e.g., Bulger et al. 1993), salinity plays a major role in the distributions of individual species and life stages; anadromous, marine-resident, estuarine, and freshwater-resident assemblages are all represented. In general, introduced species are most abundant in the freshwater-resident assemblage (Feyrer and Healey 2003; Nobriga et al. 2005)

THE PELAGIC ORGANISM DECLINE ("POD")

The Interagency Ecological Program (IEP), a consortium of nine state and federal agencies, has been monitoring fish populations in the San Francisco Estuary for decades, and has developed one of the longest and most comprehensive data records on estuarine fishes in the world. One of the most widely-used IEP databases is fish catch from the fall midwater trawl survey, which has been regularly conducted by DFG since 1967 (Stevens and Miller 1983; Sommer et al. 1997). This survey samples the pelagic fish assemblage in the upper estuary, the tidal freshwater and brackish portion of the system from the delta to San Pablo Bay. The most abundant resident pelagic fishes captured are two native species, delta smelt (*Hypomesus transpacificus*; Figure 3) and longfin smelt (*Spirinchus thaleichthys*), and two introduced species, striped bass (*Morone saxatilis*) and threadfin shad (*Dorosoma petenense*).

The San Francisco Estuary is physically very dynamic, so it is not surprising that annual abundance of all of these populations is extremely variable (Figure 4), and that much of this variability is associated with hydrology (Figure 2). Historically, the lowest abundance levels for the pelagic fishes typically have occurred in dry years, such as a six-year drought during 1987–1992. Consistent with this observation, several of these species show strong statistical associations with flow during their early life stages (Stevens and Miller 1983; Jassby et al. 1995; Kimmerer 2002a).

As some of the leading scientists in the IEP, we became concerned when fall midwater trawl abundance indices for these four pelagic fishes began to decline around 2000 (Figure 4). The situation deteriorated over the next several years. Abundance indices for 2002-2005 included record lows for delta smelt and young-of-the-year striped bass, and near-record lows for longfin smelt and threadfin shad. By 2004, these declines became widely recognized and discussed as a serious issue, and collectively became known as the Pelagic Organism Decline (POD).

The extreme variability in the data makes it difficult to say whether these indices are truly at unprecedented low levels. Mean catch per trawl with 95%

Figure 3. Adult delta smelt, a federally-listed species whose range overlaps with diversions that supply water for over 25 million people.

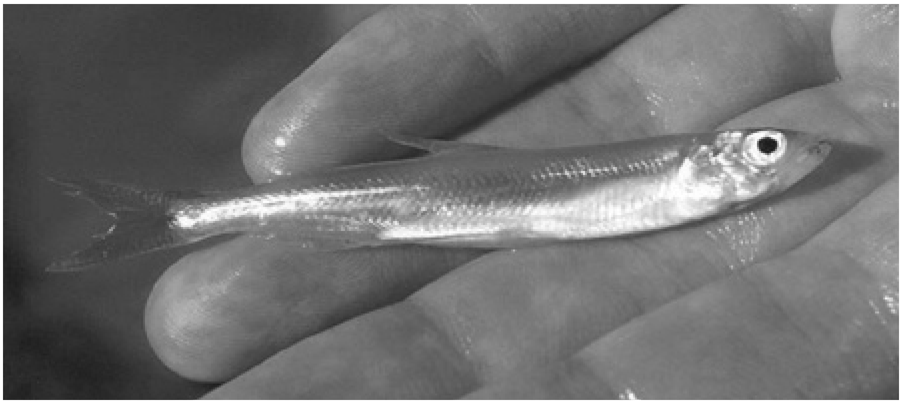
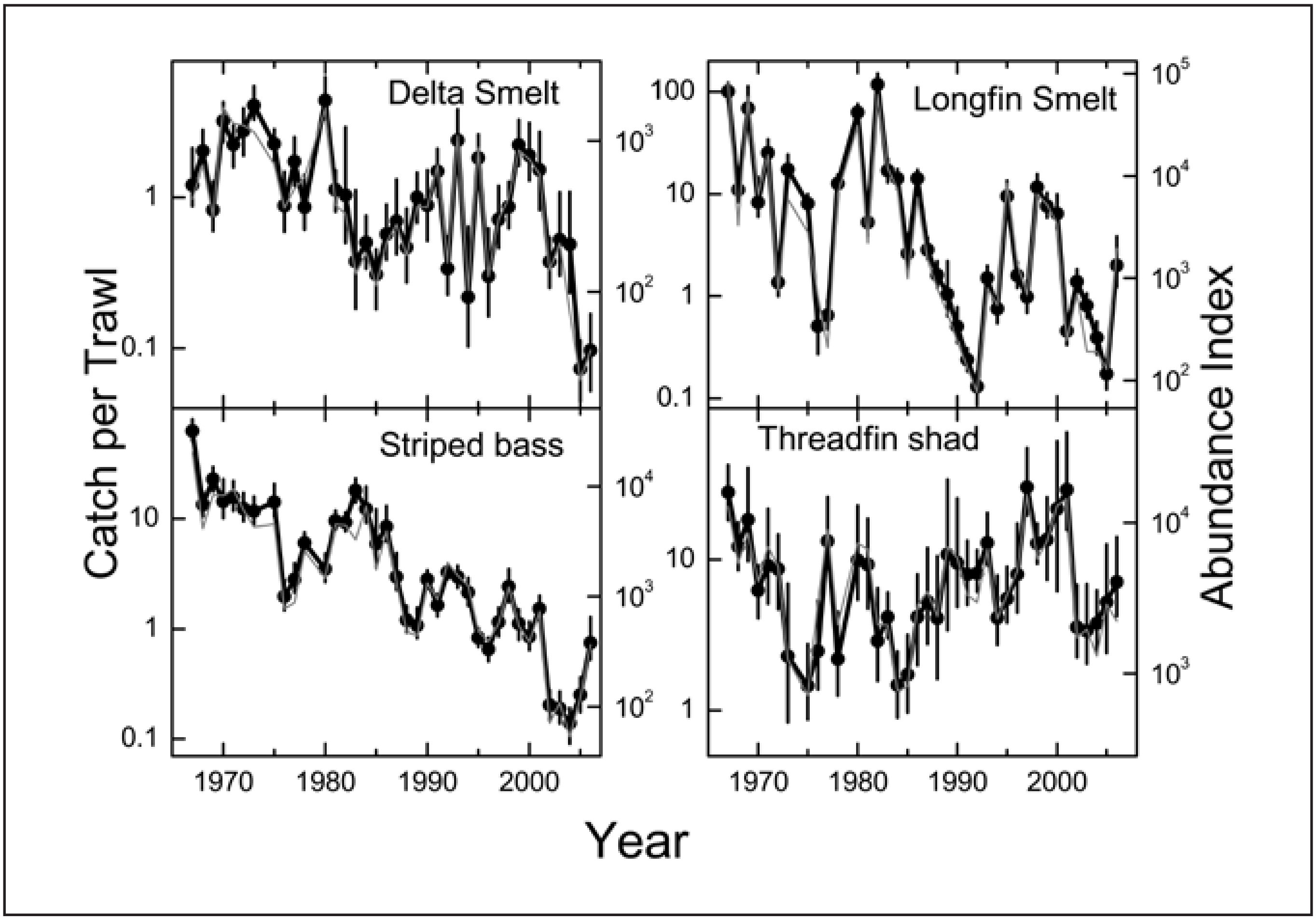
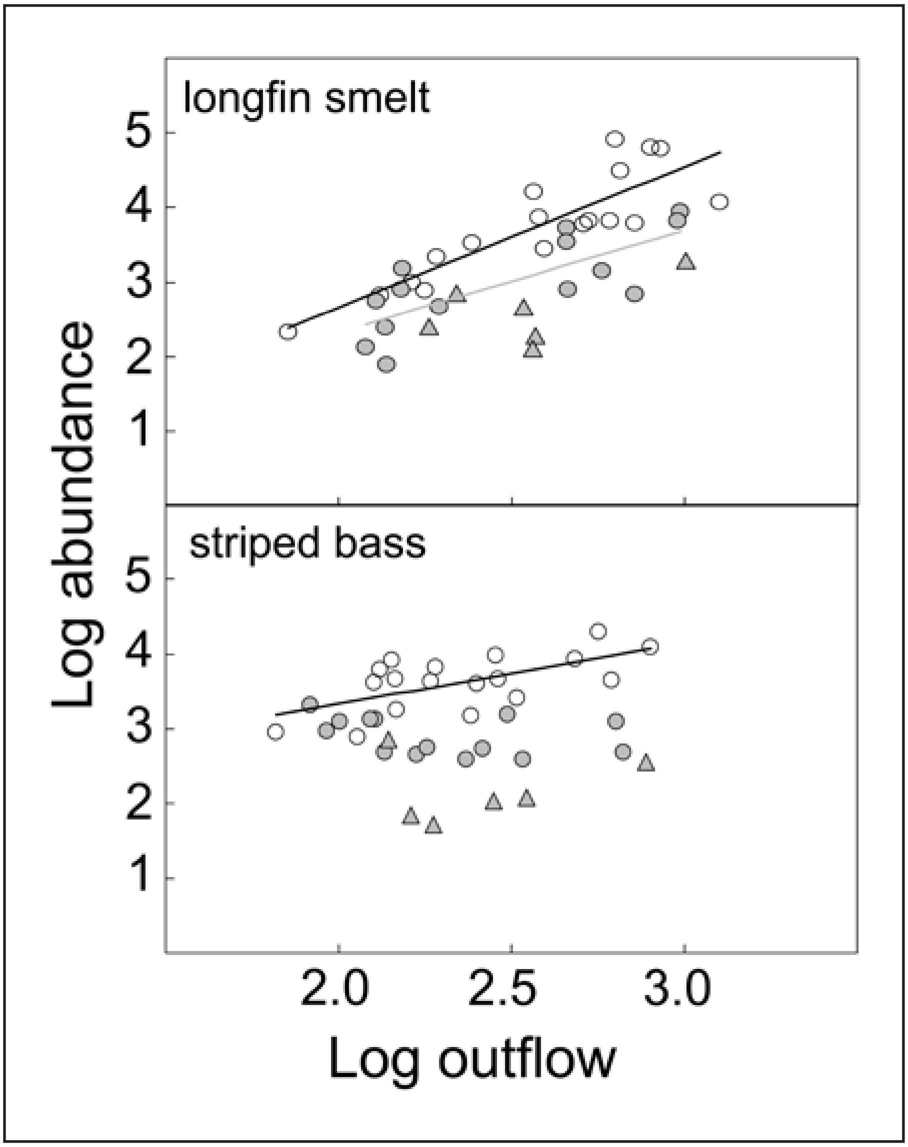


Figure 4. Trends in four pelagic fishes during 1967–2006 based on the fall midwater trawl, a DFG survey that samples the upper San Francisco estuary. Symbols with heavy lines and error bars (left y axis) show mean catch per trawl (all stations) with approximate 95% confidence intervals determined by bootstrap analysis (Kimmerer and Nobriga 2005), and the thin lines (right y-axis) show abundance indices. No sampling occurred in 1974 or 1979. Development of abundance indices from catch data is described by Stevens and Miller (1983). Note that the y-axes are on logarithmic scales.



confidence intervals developed using resampling methods indicate that the recent indices are indeed quite low, and for some species the lowest on record (Figure 4). Abundance improved somewhat for each species during 2006, but the levels remain relatively poor as compared to long-term trends. Moreover, these low abundance levels are remarkable in that winter and spring river flows into the estuary were moderate or very wet (2006) during the recent years (Figure 2), conditions that typically result in at least modest recruitment of most of the pelagic fishes. Longfin smelt is perhaps the best example of this point as the species shows a very strong relationship with delta outflow (Figure 5). The introduction of the overbite clam (*Corbula amurensis*) in 1986 and associated changes in the food web reduced the magnitude of the response of longfin smelt without altering its slope (Kimmerer 2002b). Specifically, the grazing effects from *Corbula* are thought to have resulted in a substantial decline in phytoplankton and calanoid copepods, the primary prey of early life stages of pelagic fishes. As a consequence, comparable levels of flow did not generate the expected levels of fish biomass (as indexed by abundance) after 1986. During the POD years, the abundance indices for longfin smelt deviated substantially downward from both the pre- and post-*Corbula* relationships with outflow. The situation is similar for young-of-the-year striped bass, whose historical association with outflow was also altered by *Corbula*, and apparently again during the POD years, when abundance indices were well below the original relationship with outflow. Hence, it appears that the response of these pelagic fishes to environmental conditions has fundamentally changed.

Figure 5. Log-log relationships between fall midwater trawl abundance indices and delta outflow for longfin smelt and young-of-the-year striped bass. Delta outflow values represent the mean levels (m^3/s) during January–June for longfin smelt, and during April–July for striped bass. The data are compared for pre-*Corbula* invasion years (1967–1987; white circles), post-*Corbula* invasion (1988–2000; dark circles), and during the POD years (2001–2006; triangles). Fitted lines indicate linear regression relationships that are statistically significant at the $P < 0.05$ level.



MANAGEMENT IMPLICATIONS OF THE POD

Delta smelt has been listed as a threatened species since 1993 under the federal Endangered Species Act (ESA; Bennett 2005). The geographic range of delta smelt is relatively narrow, and overlaps with the SWP and the CVP diversions (Figure 1) which supply water to over 25 million people in the state and to over 500,000 ha of farmland in the San Joaquin Valley alone, supporting a multi-billion dollar agriculture industry. Moreover, the delta smelt is primarily an annual species, so multiple age classes are not available to buffer the population against environmental catastrophes. As a consequence, for many years the species has been the focus of a wide range of protective management actions. Each year, decisions about water use costing millions of dollars are affected by the status and distribution of delta smelt. Much of the effort to improve the delta smelt population has been led by CALFED, an interagency group formed largely because of long-term declines in delta smelt and other native fishes (Koehler 1995). To help improve the status of delta smelt and other native fishes, the CALFED effort invested $335 million in over 300 habitat restoration projects through 2002, and developed a large allocation of water for use by fisheries agencies, the Environmental Water Account (CALFED 2003). Note, however, that only a portion of these actions have been focused directly towards pelagic fishes.

Among the numerous consequences of the recent low abundance indices was a March 2006 petition by environmental groups to change the federal and state listing status of delta smelt from "threatened" to "endangered" based on the argument that its extinction risk has increased. The collapse of the delta smelt population and the other pelagic fishes also resulted in a U.S. Fish and Wildlife Service ESA reconsultation (Section 7) for the operation of the SWP and CVP diversions, several lawsuits filed against the water projects, numerous front-page newspaper articles, and hearings by the U.S. Congress and the California legislature. As of the writing of this article, the SWP is under court order to cease water diversions within 60 days unless a California Endangered Species Act permit is obtained to cover incidental take of delta smelt and other listed species. The principal outcome of all this activity is substantial uncertainty about the reliability of the state's water supply.

THE POD INVESTIGATION

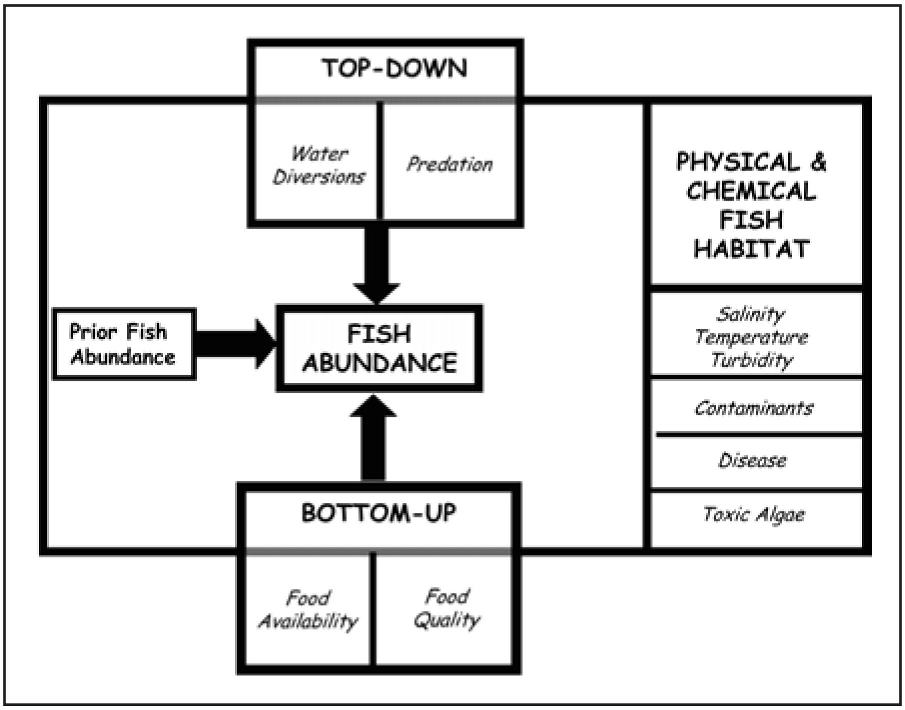
In response to the POD, the IEP formed a work team in 2005 to evaluate the potential causes of the decline (IEP 2005, 2006). The team organized an interdisciplinary, multi-agency effort including staff from DFG, California Department of Water Resources, Central Valley Regional Water Quality Control Board, U.S. Bureau of Reclamation, U.S. Environmental Protection Agency, U.S. Geological Survey, CALFED, San Francisco State University, and the University of California at Davis. A suite of 47 studies was selected based on the ability of each project to evaluate the likely mechanisms for the POD, and the feasibility of each project in terms of methods, staffing, costs, timing, and data availability. In addition to funding for regular IEP monitoring, the program's budget was augmented by $2.4 million in 2005, and $3.7 million each for 2006 and 2007 to fund the recommended research. Because of the high profile of the POD study, the team has committed to an unusually high level of outreach to agencies, the public, and the scientific community.

The POD study is organized around a relatively simple conceptual model to describe possible mechanisms by which a combination of long-term and recent changes in the ecosystem could produce the observed pelagic fish declines (Figure 6). This conceptual model is rooted in classical food web and fisheries ecology and contains four major components: (1) prior fish abundance, which posits that continued low abundance of adults leads to reduced juvenile production (i.e., stock-recruitment effects); (2) habitat, which posits that estuarine water quality variables, disease, and toxic algal blooms affect estuarine species; (3) top-down effects, which posits that predation and water project entrainment affect mortality rates; and (4) bottom-up effects, which posits that food web interactions in Suisun Bay and the west delta have limited fish abundance. For each model component, our working hypotheses are that the component was responsible for an adverse change at the time of the POD and that this change resulted in a population-level effect.

The first model component, prior adult abundance, is based on the expected influences of stock-recruitment effects. At least weak stock-recruitment effects have been reported for delta smelt (Bennett 2005), although environmental factors are thought to dominate at most abundance levels. Recent habitat changes (Model Component 2) include shifts in flow patterns, largely a consequence of upstream dam operations that have resulted in lower winter and spring inflow and higher summer inflow to the delta (Kimmerer 2002b), and fall salinity encroachment (Feyrer et al. 2007). Changes in habitat include basic water quality variables such as salinity, turbidity, and temperature. In addition, a broad suite of herbicides and insecticides are applied throughout the watershed, which can result in toxicity to fish and their prey (Werner et al. 2000; Kuivila and Moon 2004). Recent changes in pesticide applications include the increasing use of pyrethroids, which are highly toxic to aquatic organisms (Weston et al. 2004). Moreover, blooms of the toxic blue-green alga *Microcystis aeruginosa* have been observed in the delta since 1999 (Lehman et al. 2005).

Because large volumes of water are drawn from the estuary (Table 2; Figure 2), water diversions and inadvertent fish entrainment are among the best-studied top-down effects (Model Component 3) in the San Francisco Estuary. The diversions are known to entrain most species of fish in the upper estuary (Brown et al. 1996), and are of particular concern in dry years, when the distributions of young striped bass, delta smelt, and longfin smelt shift closer to the SWP and CVP water diversion facilities (Stevens et al. 1985; Sommer et al. 1997). As an indication of the magnitude of the effects, approximately 110 million fish were salvaged at the SWP screens and returned to the delta over a 15-year period (Brown et al. 1996). However, this estimate does not include other substantial effects including mortality of fish in the waterways leading to the diversion facilities, losses of larvae <20 mm FL that are not collected on fish screens, and losses at the CVP. The effects of predation are less well-understood in the estuary. A recent proliferation of aquatic weeds has provided habitat resulting in a substantial increase in

Figure 6. The basic conceptual model for the pelagic organism decline (POD).



inshore predators such as centrarchid fishes (Nobriga et al. 2005; Brown and Michniuk 2007). However, it is unclear whether the littoral communities have a major effect on the dynamics of pelagic fishes.

The last model component, bottom-up effects, also has received substantial attention in the estuary as a consequence of the extreme level of species introductions, resulting in major changes in the pelagic food web (Cohen and Carlton 1998). Phytoplankton biomass (as indexed by chlorophyll *a*) has declined over the last 4 decades, and species composition has shifted, with a sharp decline in diatom abundance and production in Suisun Bay and the western delta (Lehman 2002; Jassby et al 2002; Kimmerer 2005). Key groups of zooplankton have likewise declined in abundance and biomass, with sharpest changes among calanoid copepods, a primary prey for early life stages of pelagic fishes (Kimmerer and Orsi 1996; Kimmerer 2006).

CONCLUSIONS

Unlike the collapses of commercial fisheries for Pacific salmon (*Onchorhynchus* spp.) or Atlantic cod (*Gadus morhua*), the POD involves an entire fish assemblage, including rare native species as well as some of the most abundant introduced species in the estuary. As such, it has focused attention not only on traditional fishery management concerns such as harvest and water management, but has led to new ecological studies of water quality and several synergistic processes.

Fortunately, the San Francisco Estuary has an exceptionally long and detailed history of environmental monitoring. The collapse required an integrated research program to analyze the problem. Analysis of the historical data, coupled with an intensive program of sharply focused studies, has permitted the rapid development of a better understanding of factors that have affected fish abundance in both the short and long term. This multi-faceted approach should greatly assist in planning for aquatic resource protection from increasing human demands and other stressors such as global warming and the imminent invasion by quagga mussels.

The scope of the POD investigations has also generated high expectations for "real-time" reporting and interpretation of the results. The pressing need to reverse the decline has also led to demands for specific practical actions to remediate problems that we only understand broadly. Management actions based on incompletely integrated results run the risk of ineffectiveness (Hutchings et al. 1997). Although the available data have allowed us to generate a conceptual model of the major factors, the individual and cumulative importance of the stressors remains unclear. Hence, effective management actions to reverse the POD will require quantitative models that can integrate the effects of multiple stressors and more detailed investigations into the causes and mechanisms of the declines. Moreover, management actions will be most useful if they can be implemented using an adaptive approach that allows fisheries scientists and resource managers to learn from designed manipulations of the upper San Francisco estuary. Such actions are currently being considered as part of the approach to deal with the POD ♋.

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FEATURE: FISH HABITAT

Streambank Restoration Effectiveness: Lessons Learned from a Comparative Study

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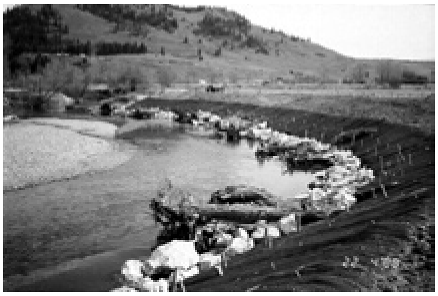
ABSTRACT: Post-treatment effectiveness monitoring should be an integral part of stream restoration efforts, but it is often neglected due to lack of funds or insufficient project planning. Here we report results of an effectiveness evaluation of a streambank restoration program for salmon streams in the southern interior of British Columbia. Restoration involved treating eroding riverbanks with bank grading, riparian plantings, and installation of rock toes, rock-wood current deflectors, and livestock exclusion fencing. Absence of pre-treatment site characterization data necessitated comparing post treatment conditions at treated sites to conditions at untreated eroding control sites. We measured in-channel and riparian conditions plus invertebrate abundance and biomass at 16 sites treated between 1997 and 2002 and 11 nearby control sites. Treatment and control sites did not substantively differ in their habitat condition or aquatic macroinvertebrate abundances, although treated sites tended to have more shrubs along the outside bank, higher inside banks, and narrower wetted widths. Absence of statistical differences between treatment and control sites might be due to low statistical power, as >50 sites per group would need to be sampled for power to reach 0.8 at the effect sizes observed. Site specific channel gradient, a variable unaffected by restoration actions, was correlated with many of the variables we measured to characterize habitat condition, thereby confounding our ability to determine the magnitude of change relating to treatment efforts. Our results demonstrate the weaknesses of relying on a post-treatment, between-group comparison experimental design for restoration effectiveness monitoring. We suggest collection of pre-treatment data should be an essential part of the restoration process so more appropriate "before-after" experimental designs can be applied.

Efectividad de la Restauración de los Márgenes de Cuencas Fluviales: Lecciones Aprendidas de un Estudio Comparativo

RESUMEN: El monitoreo de la efectividad pos-tratamiento debiera ser parte integral de la restauración de cuencas fluviales, sin embargo éste no siempre se toma en cuenta ya sea por una planeación insuficiente o a la falta de financiamiento. Aquí se reportan los resultados de un programa de evaluación de la efectividad de la restauración de los márgenes de cuencas fluviales en la porción sur del interior de la Columbia Británica. La restauración implica la erosión de la cuenca por dragado, plantaciones ribereñas y la instalación de escolleras, deflectores de corrientes hechos de madera y roca, y cercado para exclusión de ganado. Ya que no existen datos sobre la caracterización de los sitios antes del tratamiento, se realizó un análisis comparativo entre las condiciones pos-tratamiento en los sitios tratados y las de los no tratados. Entre 1997 y 2002 se midieron las condiciones de la cuenca, la zona ribereña y la abundancia y biomasa de invertebrados en 16 sitios tratados y en 11 sitios de control. No hubo diferencias notables entre ambos sitios en cuanto a las condiciones del hábitat o la abundancia de macro-invertebrados, sin embargo los sitios tratados presentaron mayor cantidad de arbustos a lo largo de la batiente erosiva del río, batientes de acarreo más elevadas y una menor cota de agua. La ausencia de diferencias estadísticas significativas entre los sitios de control y los tratados pudo ser debida al bajo poder estadístico ya que, dada la magnitud de los efectos observados, se necesitarían muestrear > 50 sitios por grupo para que el poder estadístico alcanzara 0.8. El gradiente del canal en cada sitio, variable no afectada por las actividades de restauración, se correlacionó con varias de las variables que caracterizan la condición del hábitat, lo que afectó la capacidad para medir la magnitud del cambio asociado a los tratamientos. Nuestros resultados demuestran las debilidades que tiene el basarse en un diseño comparativo pos-tratamiento para el monitoreo de la efectividad de la restauración. Sugerimos que, mientras no puedan aplicarse diseños experimentales de condiciones "antes y después", la colecta de datos pre-tratamiento debiera ser una parte esencial en el proceso de restauración.



INTRODUCTION

Habitat management, including habitat restoration, entails applying one or more treatments and should be viewed as an experiment which necessitates post-treatment evaluation (Kondolf and Micheli 1995; Kershner 1997; Michener 1997; Palmer et al. 2005; Stem et al. 2005; Woolsey et al. 2007). Michener (1997) suggested the theoretical optimum for restoration effectiveness monitoring as "long-term monitoring of salient patterns and processes in adequately replicated control and experimental units at appropriate spatial and temporal scales using sound sampling design and statistical analyses." However, Michener concedes this optimum is rarely achieved and often unachievable. Unfortunately, any amount of systematic monitoring of the results of freshwater habitat management efforts remains an exception, not the rule (Kondolf 1998; Pretty et al. 2003; Quigley and Harper 2006; Reeve et al. 2006).

Limiting factors precluding efficient post-treatment evaluation often originate from insufficient pre-project planning. For example, many projects fail to incorporate effectiveness monitoring into the initial project budget and evaluation is therefore abandoned due to lack of funds (Reeve et al. 2006). Similarly, restoration practitioners often fail to provide a clear statement of project goals, and therefore effectiveness monitoring has no criteria on which to judge project success or failure (Kondolf 1995; Palmer et al. 2005; Stem et al. 2005). In other cases, projects fail to collect appropriate pre-treatment data, which precludes a before-after experimental design or its derivatives such as before-after-control-impact (BACI; Green 1979; Walters et al. 1988; Roni et al. 2005) and typically forces reliance on less powerful post-treatment between-group comparisons (Mellina and Hinch 1995; Bryant et al. 2004).

Beginning in the 1990s, the Habitat Management Unit for the Southern Interior of British Columbia of Fisheries and Oceans Canada and its local partners initiated an eroding streambank restoration program for tributaries of the Thompson River system. The three explicitly stated goals were to stop bank erosion, increase native salmonid production, and foster a stewardship mentality within the local community. Between 1992 and 2005, >200 eroding banks, spread across 5 valley floor mainstem rivers, had been treated. By 1997, largely via learning from past structural failures, treatment methods had evolved to a standard template involving bank grading, riparian plantings with willow (*Salix* spp.) cuttings, livestock exclusion fencing, and installation of a rock toe coupled with site specific mixtures of tree and/or rock current deflectors, bank contouring, and occasional plantings of deciduous (primarily *Populus balsamifera* and *Betula papyrifera*) and/or coniferous (primarily *Pseudotsuga menziesii*) trees. Pre-treatment data characterizing site conditions were not collected at any of the treated sites.

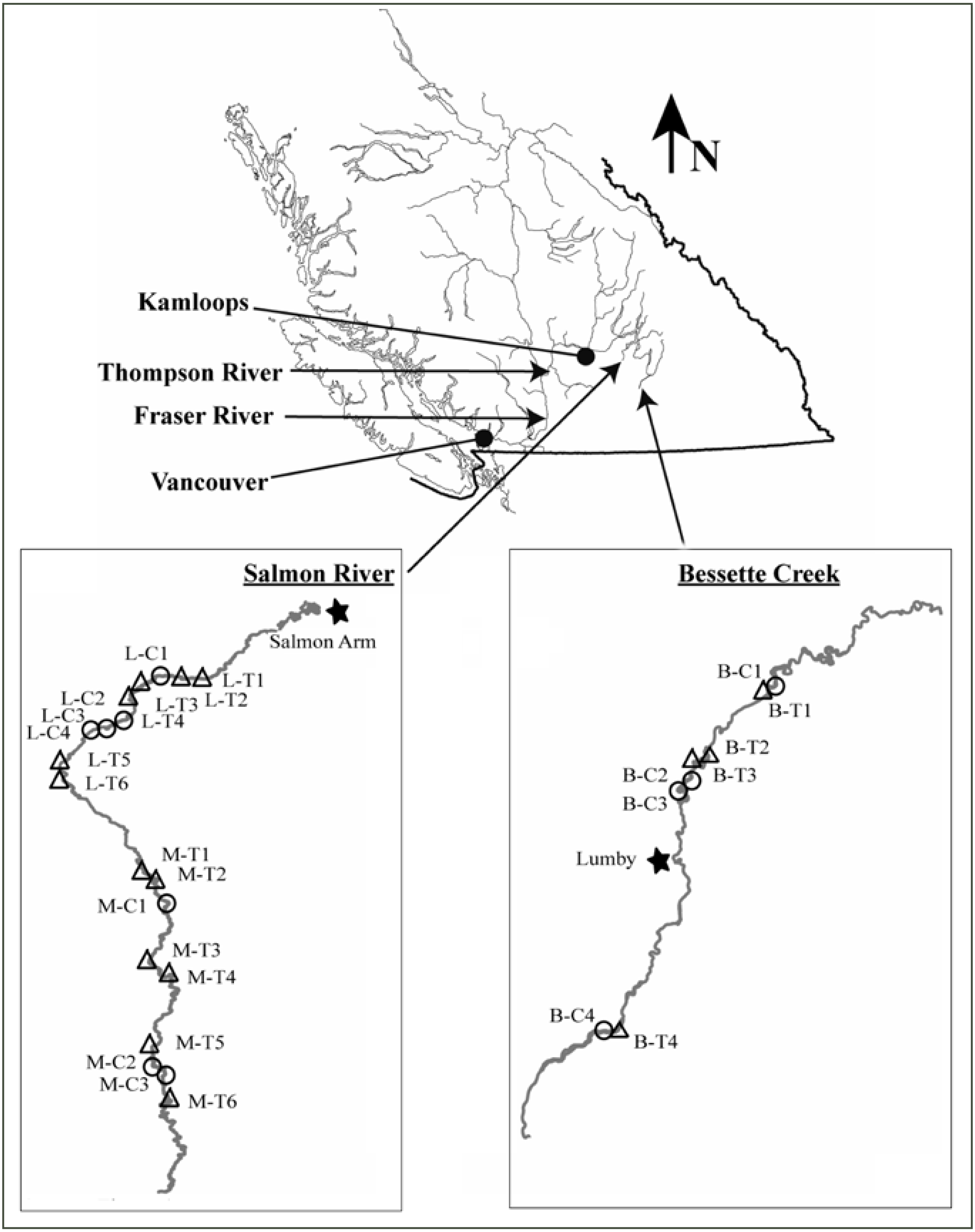
A visual survey of project structural integrity conducted in 2005 found that all of the 81 streambank restoration projects constructed along the Salmon River since 1997 had structural integrity ratings of "adequate" or better, equating to no evidence of physical failure, and that all were accomplishing their proximal goal of erosion control (S. Bennett, unpublished data). Although not explicitly quantified, structural and functional integrity of similar projects in nearby watersheds, including the approximately 20 Bessette Creek projects completed to date, also appeared to be consistently good (M. Cooperman, pers. observ.). None of the structural integrity assessments evaluated ecological effects of the bank restoration efforts.

In this article, we report results of an extensive post-treatment effectiveness evaluation of streambank restoration efforts in the Salmon River and Bessette Creek. We compare stream channel and riparian vegetation condition and aquatic invertebrate abundance and biomass at 16 sites "treated" between 1997 to 2002 to those at 11 actively eroding "control" sites. We hypothesize that relative to the control sites, treated sites would have greater in-channel habitat diversity, higher depth-to-width ratio, larger streambed mean particle size, greater riparian zone plant coverage on both banks of the channel, greater amounts of natural vegetation recruitment on point bars opposite treated banks, and greater aquatic macro-invertebrate abundances. Because our assessment was based on between-site comparisons, not before-after comparisons of individual treated sites, we also evaluated the nature of site-to-site variability and how this variability related to, and potentially influenced, site-specific conditions and response to restoration. We discuss our results in the context of the limitations of an extensive post-treatment experiment design for effectiveness monitoring and provide suggestions for future restoration monitoring efforts.

STUDY SITES

The Salmon River and Bessette Creek occupy the "interior Douglas fir—very hot—dry" biogeoclimatic zone of British Columbia (Lloyd et al. 1990), and drain to the Pacific Ocean via the Thompson River sub-basin of the Fraser River watershed (Figure 1). Valley floor elevations range between 350–500 m above sea level, annual mean precipitation is 400–500 mm, and soils consist of a blanket of poorly sorted moraine deposits within a matrix of sand-silt-clay with limited fluvial reworking (Lloyd et al. 1990). Timber harvest and irrigated agriculture-ranching are dominant land uses in both watersheds and almost all valley floor land is privately held in agriculture. In 2002, the Salmon River experienced the third highest peak discharge of the 31-year period of record (49.2 m^3/s; Water Survey of Canada station 08LE021) and Bessette Creek experienced the ninth highest peak discharge

Figure 1. The top map shows the Fraser River watershed in the lower half of British Columbia, Canada (scale 1 cm = 125 km) and the location of the Salmon River and Bessette Creek in the headwaters of the Thompson River sub-basin. The lower maps show the distribution of study sites along the two rivers. Scale for the Salmon River map is 1:100,000 and for Bessette Creek 1:50,000. Treatment sites are triangles and control sites are open circles. Study site nomenclature is described in the text..



in its 32-year period of record (32.3 m^3/s; WSC station 08LC042), indicating all treatment sites included in our study had experienced a high discharge event post-treatment and therefore had the potential to display a geomorphic response to treatment. Mean discharge during our field effort was 2.09 m^3/s in Salmon River and 2.50 m^3/s in Bessette Creek.

Miles (1995) estimated >40% of the forest cover of the Salmon River watershed has been harvested since the early 1900s, that approximately 20% of the mainstem was actively eroding, that the channel ranged from 11–211% wider than it was in the 1930s, and that in the lower 60 km of the river (the area where our study occurred) 50% of the channel had either no riparian vegetation or a riparian band less than 1 channel width wide, in contrast to abundant and well-dispersed riparian-gallery vegetation of the 1930s. Quantitative land use and impact data for the Bessette Creek watershed are not available but are assumed to be similar to those of the Salmon River owing to the close proximity of the two watersheds and similarities in general land use patterns. Sediment supply and movement through these two systems have not been studied, but are assumed to be high owing to numerous eroding banks and the rapidity with which constructed in-channel habitat structures are buried (S. Bennett, pers. observ.).

METHODS

Study site selection

Restoration activities have been conducted on about 100 eroding banks of the Salmon River and on approximately 20 eroding banks along Bessette Creek since the mid-1990s. To qualify as a treatment site in our study, the following was needed: (i) restoration activities occurred between 1997–2002 inclusive, (ii) restoration activities involved the outside bank of a channel meander of the mainstem channel on the floor of an unbound alluvial valley segment, (iii) <5% of the length of the as-built project could displayed evidence of physical failure (failure of riparian plantings was not included in this physical failure criterion), (iv) site-specific actions must have successfully accomplished the proximal goal of halting bank erosion, and (v) the site must not be directly influenced by civil engineering works, tributary inputs, other site-specific restoration activities that were unrelated to bank restoration (e.g., artificial riffles or channel re-configuration), or possess unique geological features such as local clay lenses or other erosion resistant inclusions. For inclusion as a control site, a riverbank needed to satisfy all applicable criteria above plus be actively eroding as evidenced by the face of the bank being unvegetated, near perpendicular to the water surface, and displaying signs of recent bank slumping. By limiting our study scope to only sites satisfying the above conditions, we are confident our control sites are a fair representation of pretreatment conditions at the treated sites.

The technical coordinators of the Salmon River and Bessette Creek watershed roundtables identified 16 restored sites meeting our inclusion criteria. We located 11 eroding bank sites in proximity to the 16 treatment sites to serve as controls (Figure 1). For identification purposes, we assigned each site a unique alpha-numeric identification based on the first letter of the river segment it occurred within (L = lower Salmon River, M = middle Salmon River, B = Bessette Creek), a T or C for treatment or control, and a number. Although numbers were assigned sequentially upstream to downstream, sites were sampled in random order. Because treatment sites were located haphazardly along the river corridors it was impossible for us to assure either random or systematic random distribution of sites. Similarly, because each treatment site received a unique treatment prescription, it was impossible to have sufficient replicates within each treatment type to allow for evaluation based on groupings of specific restoration techniques. However, four treatment sites (L-T5, M-T2, B-T3, B-T4) had received notably more comprehensive restoration than other treated sites, including self-launching rock spurs and trenched rock toes coupled with multiple outward facing >1.5 m diameter root wad revetments embedded into a sloped bank with rock groins at the upstream and downstream ends of the treatments, allowing use to compare conditions at these four "intensive" treatment sites to conditions at the other 12 treatment sites.

At each location, the study "bank" was the portion of the outside meander bend that had either received restoration or was actively eroding. Each study bank occurred within a study "reach," defined as the portion of the channel lying between the upstream and downstream thalwag cross-over points bracketing a study bank. Figure 2 depicts how we organized study sites.

Channel condition

We conducted a habitat unit survey for each study reach following Bisson and Montgomery (1996). Length of a habitat unit was the longest axis, width was the widest point perpendicular to length, and depth was the deepest point. We pooled riffles and runs into "fast water," classified pools as "slow water," left glides as its own category, and determined proportion of a reach's total surface area within each habitat class. The proportion of each reach classified as fast and slow water was highly correlated (r = -.87, n = 27, P <0.001), so surface area as fast water was eliminated from subsequent analyses.

We followed Harrelson et al. (1994) to develop elevation profiles along each transect. Elevation surveys extended from 2 m outside the top of bank on each side of the channel and elevation was recorded at every 0.5 m along each transect with supplemental readings taken at top of bank, bottom of bank, and water's edge. We assumed depth equaled 2.0 m at any point too deep for safe surveying. Mean depth of a reach was the mean of all individual water depths.

We determined bankfull, active channel, and wetted widths along each transect. Active channel width was from top of the outside bank to the top of the first distinct slope change along the point bar of the inside bank. Owing to the long history of channel widening in these systems, along with our observations of large accumulations of living and dead plant material above this slope break, we interpreted the land between the slope break and the top of inside bank to be "incipient floodplain," which we considered part of the riparian portion of the fluvial system. Depth-to-width ratio of a reach was derived from mean reach depth and mean wetted width. We used wetted width because summer time water temperatures are a primary management concern and therefore it is the wetted portion of channel at base flow condition that is the variable of concern. Mean depth was significantly correlated with depth-to-width ratio (r = 0.90, n = 27, P <0.001), so mean depth was eliminated from subsequent analyses. Height of the inside bank at 1 m from the water's edge was the mean of transect specific differences in elevation between the high point within 1 m of the water's edge along the inside bank and the elevation of the water's edge on the inside bank.

We visually estimated the proportion of stream bed particles falling within particle size classes as in Harrelson et al. (1994; organic matter, silt, sand, gravel, small or large pebbles, small or large cobble, boulder) within an approximately 1 m^2 portion of the stream bed underlying the thalwag and along the inside bank point bar of each reach (Figure 2). We pooled gravel and smaller particles (β-axis <4 mm) into the category of "fine sediments" and all small pebbles and larger particles (β-axis >4 mm) into "coarse sediments" and determined the proportion of fine and coarse sediments within the two sites per reach. We limited subsequent analyses to proportion of fine sediments in point bars and proportion coarse sediments underlying the thalwag. Sediment data were not collected at L-T6 or M-T5.

Channel gradient was the difference in elevation between the upstream and downstream ends of a study reach, measured at the water's edge, divided by the distance between the two points as measured along the curvature of the water's edge along the inside bank.

Riparian assessment

We used a modified line intercept technique (McDonald 1980) along the same transect lines as the elevation survey to assess the coverage of riparian vegetation on both the inside and outside bank of each study reach (Figure 2). On the outside bank of the channel, vegetation survey started at the top of bank. On the inside bend, the vegetation survey began at the edge of the active channel. Riparian surveys extended 5 m from start points, were 1 m wide (0.5 m on each side of the transect line), and assumed to reach indefinitely upwards. For each tree or shrub of which any part of the plant entered the survey plane, we recorded the species and the length of the portion of plant within the survey plane. For the inside bank, we also tallied the number of seedlings (trees <0.5 m height) along the transect. We pooled the vegetation data from all transects on one side of the channel and determined proportional coverage by trees and shrubs, individually and combined. For example, in a case of a reach with 3 transects, the proportion outside bank covered by trees equaled the sum of the lengths of trees entering the survey planes of the 3 transects on the outside bank divided by 15 m (e.g., 3 transects, each 5 m long). For simplicity, we refer to our proportional coverage data as "coverage."

Channel condition and riparian assessments were conducted in July and August 2005, when discharge in the study systems had stabilized to summer baseflow, thereby limiting the influence of falling discharge on flow dependant values such as amount of habitat as fast or slow water. Whenever judgment was used during data collection (i.e., habitat unit survey, bankfull dimensions, sediment particle sizes), all determinations were exclusively made by the senior author, thereby standardizing measurement precision and eliminating among-crew estimation bias (Woodsmith et al. 2005).

Aquatic macroinvertebrates

During the first two weeks of September 2005, we revisited each study reach to collect six benthic macroinvertebrate samples using a Surber sampler (400 *u*m mesh, 0.5 m^2 quadrant size). We divided the downstream portion of the inside bank point bar at 1 m from the water's edge (e.g., same area as the point bar sediment surveys; Figure 2) into ten 1-m long intervals and used a random number table to select 6 of the 10 units for sampling. Samples were collected by agitating the surface substrate and all larger rocks and wood within the quadrant for one minute and samples were field preserved in 10% buffered formalin. We also collected 1 three-minute kick net sample per site (400 *u*m mesh) by moving the kick net across the channel in a upstream progressing zig-zag starting immediately upstream of the Surber sample locations. Because kick net and Surber samples support similar conclusions (Cooperman et al. 2006), we report only Surber sample results.

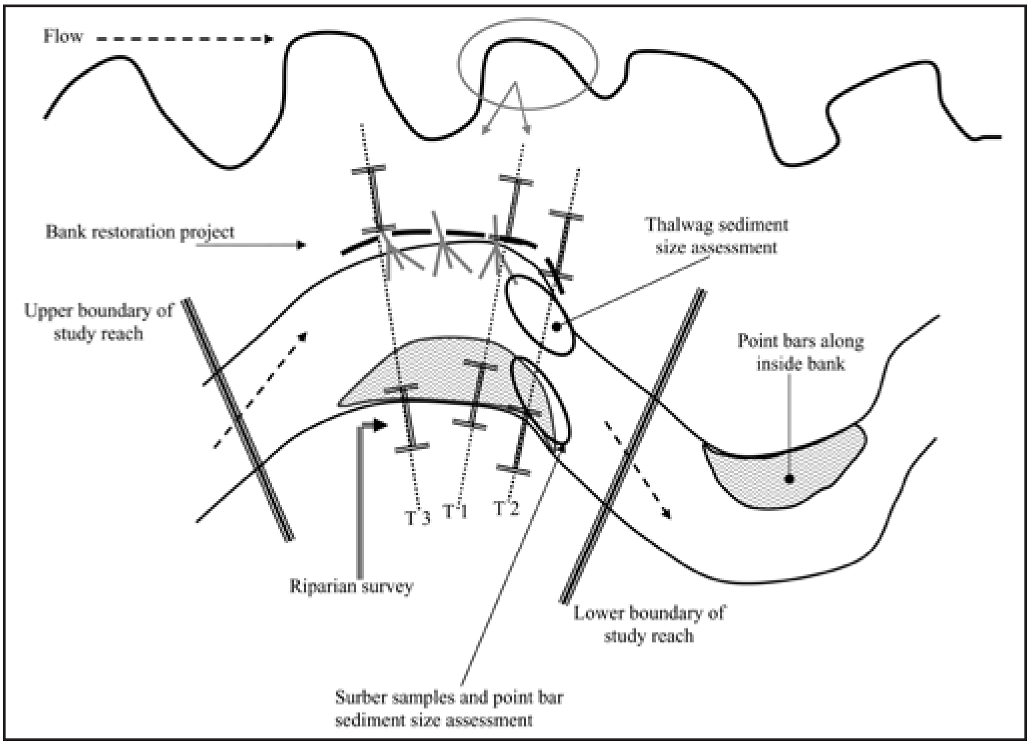


Figure 2. Schematic representation at a treatment site depicting location of three transects used for channel elevation and riparian surveys, upper and lower reach boundaries, and sediment and invertebrate sampling locations. Length of each study bank was that portion of the bank which had either received active restoration or was actively eroding. We established 3 transects for study banks up to 100 m long and 5 for banks >100 m. Transects extended across the active channel perpendicular to the thalwag. Transect 1 was always positioned at the point of maximum curvature and transects 2 and 3 were at 10% of the bank length inside the downstream and upstream ends of the study bank respectively. When applicable, transects 4 and 5 were halfway between transects 1 and 2 and 1 and 3 respectively. One control bank (B-C2) had the thalwag inflection point at the downstream end of the bank and therefore only transects 1 and 3 were established. Diagram is not to scale.

In the laboratory, we randomly selected two of the six Surber samples from each site for analysis. We picked all invertebrates under 3x magnification, identified them to order, and used the mean of the two samples per site to determine site-specific total number of macroinvertebrates, number of ephemoptera-plecoptera-trichoptera (EPT), total macroinvertebrate wet mass, and EPT wet mass. Because all possible pairings of the four response variables were significantly correlated (all P values <0.0433, n = 27), only total macroinvertebrate abundance was used in subsequent analyses.

DATA ANALYSIS

Preliminary results indicated that analyzing study sites of the three river segments of Bessette Creek, lower Salmon River, and middle Salmon River separately for differences between treatment and control sites supported the same conclusions as analyses of all segments pooled into a single population (Cooperman, unpublished data). Therefore we only report results stemming from the pooled data set.

Assessing effects of restoration activities

We used multivariate analysis of variance (MANOVA) as a single test for differences between treatment and control sites. The eight dependant variables entered into the MANOVA were HabUnits, P SA Slow, P SA Glide, D:W, Hgt IB, P TH Coarse, Cov T&S IB, #Recruits (see Table 1 for descriptions of these variables). We did not include outside bank plant abundances in multivariate analyses because the presence of plants on outside banks could be attributable to opposing mechanisms of successful establishment from restoration efforts or progression of an eroding bank into mature vegetation. We excluded P PB Fine from MANOVA due to a significant inverse relationship with P TH Coarse (r = -0.45, n = 27, P = 0.020).

We used one way analysis of variance (ANOVA) to facilitate interpretation of the MANOVA results and test specific predictions about differences between treatment and control sites (Table 1). In addition to treatment vs. control comparisons, we report the mean value of the four intensive treatment sites for each variable, but did not conduct statistical analyses of differences between intensive treatment sites and treatment or control sites due to small sample sizes. We followed ANOVA tests with a posteriori power analysis following Winer (1971) for each variable to determine the sample size needed to attain statistical power of 0.8 based on a one-tailed test at α = 0.1 and an effect size equal to the observed difference between treatment and control sites.

To test for differences in the erosion-deposition environments between treatment and control sites as a function of local channel gradient, we compared the slopes of regression lines resulting from independently regressing treatment and control site P TH coarse on gradient, and we repeated the treatment-control comparisons for the regression of P PB Fines on gradient. We tested the regression solutions for unusually influential points based on Cook's distance and DEFITS values and eliminated offending cases as needed.

We assessed differences in macroinvertebrate abundance (# Inverts) between treatment and control sites using two-way ANCOVA Analysis of Covariance with site (two levels—treatment or control) and river segment (three levels) as main effects with site specific channel gradient entered as a co-variate. We did not test for an interaction between the two main effects. To allow for posteriori power analysis for determination of statistical power of the treatment to control comparisons and to determine the desired sample sizes needed for a power of 0.8, we conducted a one-way ANOVA for differences between treatment and control sites and power analysis following the methods described above.

All statistical tests were evaluated for significance at the level of α = 0.1 owing to the higher P value criteria reducing the probability of a type II error, which can be a costly mistake in applied research given its influence on management decisions about future restoration efforts that may be based on our results (Peterman 1990).

Table 1. Variables used in data analyses, transformations applied to attain univariate normality, and abbreviations used in text and subsequent tables and figures. Log transformations are to base 10. NA = not applicable.

Group	Variable	Transformation use	Abbreviation
In-channel	Number of habitat units	NA	HabUnits
	Proportion of surface area as slow water	Arc-sine	P SA Slow
	Proportion of surface area as glide	NA	P SA Glide
	Wetted width of the active channel	NA	Width
	Depth to width ratio	NA	D:W
	Height of the inside bank at 1 m from the water's edge	Log (X)	Hgt IB
	Proportion of all sediments along the inside bank point bar in the "fine" size class	NA	P PB Fine
	Proportion of all sediments underlying the thalwag in the "coarse" size class	NA	P TH
	Gradient	NA	Gradient
Riparian	Coverage of trees on outside bank	Log (X + .01)	Cov T OB
	Coverage of shrubs on outside bank	NA	Cov S OB
	Coverage trees on inside bank	Log (X + .01)	Cov T IB
	Coverage shrubs on inside bank	Log (X + .01)	Cov S IB
	Coverage of trees and shrubs on inside bank	Log (X + 1)	Cov T&S IB
	Number of seedling trees on inside bank	Log (X + 1)	#Recruits
Aquatic macroinvertebrates	Total abundance in Surber Samples	Log (X)	#Inverts

For all ANOVAs, we tested for distributional outliers defined as values greater than 3 times the inter-quartile range from either the 25th or 75th percentile value and excluded outliers.

Characterizing site-to-site variability

We used non-metric multidimensional scaling (NMS; PC-Ord v. 4.0) to explore the nature of site-to-site variability as an aid in visualizing the relationships among measured variables. We used non-parameteric NMS ordination because it is well suited to multivariate data in which variables are measured on dissimilar scales, and because NMS is based on ranked similarities between sites it relaxes the assumption of a linear relationship between independent and dependant variables (McCune and Grace 2002) and preserves between-site distances (Clarke 1993). As such, NMS is a highly effective tool for graphical representation of community structure (Clarke 1993).

The NMS ordination incorporated all treatment and control sites ($n = 27$) and involved two data matrices. The first matrix included eight measures of habitat condition: HabUnits, P SA Slow, P SA Glide, D:W, Hgt IB, P TH Coarse, Cov T&S IB, # Recruits, and gradient. For the two cases where sediment size data were not available, we substituted the mean value of the applicable river segment—treatment group combination (i.e., P TH Coarse for L-T6 was the mean of P TH Coarse of all other lower Salmon River treatment sites). The second matrix contained two descriptive parameters: treatment or control, and river segment. Prior to ordination, we applied a general relativization by columns. No cases were identified as multivariate outliers (>2 standard deviations from the multivariate mean) and therefore both matrices had 27 cases. We used Euclidean distance and the "slow and through" autopilot setting to execute 50 runs with real data with random start configurations. We used 30 runs per tested dimension Monte Carlo simulation with randomized data to determine the number of dimensions to use in the final solution. We selected a three-dimension solution as the best fit and rotated the final solution to maximize correlation with gradient along the first axis.

RESULTS

Assessing effects of restoration activities

Treatment and control sites did not differ in multivariate space (MANOVA w/ 8 and 18 df, $F = 0.45$, F critical = 1.02, $P = 0.460$); however, treatment sites had narrower wetted widths and higher inside banks than did control sites (Table 2). All other in-channel response variables and channel gradient did not statistically differ between treatment and control sites (Table 2), although there were trends as treatment sites had greater mean number of habitat units and mean depth-to-width ratio, lower mean value of surface area as either slow water or glides, and lower mean value of fine particles along the edge of the inside bank point bars than did control sites. Differences in means were largest when treatment-to-control comparisons were limited to only the four intensive treatment sites, but the small number of intensive treatment sites precluded testing for statistical significance (Table 2). Coverage of shrubs on the outside bank of study reaches was the only riparian variable to statistically differ between treatment and control sites although treatment sites had greater mean values than control sites in all riparian categories (Table 3).

A posteriori power analysis indicated our statistical analyses of in-channel condition and riparian coverage typically suffered from low power (Tables 2 and 3). Excluding the two variables wetted width and height of inside bank, for which significant differences were found between treatment and control sites, the number of both treatment and control sites that we would have had to sample to find significant differences at $\alpha = 0.1$ and the observed effect size

Table 2. Means (± 1 standard deviation) of treatment sites ($n = 16$), control sites ($n = 11$), intensive treatment sites ($n = 4$), ANOVA P values for contrasts of treatment and control sites, and number of sites needed in both treatment and control groups to attain power of 0.8 (desired n). Means and standard deviations shown are for untransformed parameters, but statistical analyses were done on transformed data. For the two sediment variables (P PB Fine, P TH Coarse) treatment site $n = 14$. Abbreviations explained in Table 1.

Variable	Treatment	Control	*P* value	Desired *n*	Intensive
Gradient	0.00186 (.0014)	0.00232 (.0020)	0.488	105	0.00162 (.0009)
HabUnits	7.20 (3.13)	7.00 (2.24)	0.866	72	8.30 (2.40)
P SA Slow	0.11 (0.15)	0.14 (0.11)	0.410	92	0.08 (0.09)
P SA Glide	0.37 (0.25)	0.48 (0.30)	0.317	55	0.32 (0.17)
Wetted width	11.8 (2.69)	13.6 (2.07)	0.076	18	11.1 (1.70)
mean D:W	0.04 (0.01)	0.03 (0.02)	0.313*	53	0.05 (0.02)
Hgt IB	0.35 (0.22)	0.22 (0.19)	0.099	21	0.30 (0.15)
P PB Fine	0.44 (0.39)	0.57 (0.35)	0.409	75	0.41 (0.37)
P TH Coarse	0.59 (0.33)	0.59 (0.39)	0.954	17,450	0.53 (0.25)

* Note: Samples had unequal variances that could not be corrected by data transformation. Power analysis to attain "desired *n*" assumes the effect size is equal to that observed in our present comparisons. "Intensive" refers to a subset of 4 of the 16 treatment sites which received much more comprehensive restoration treatments than the other treatment sites. Mean values of treatment sites includes the four intensive treatment sites.

ranged from 53 (mean D:W) to 17,450 (P TH Coarse), with a median of 92 (Table 2).

Sediment size distribution was related to channel gradient but thalwag sediments and point bar sediments differed in response to treatment (Figure 3). Within both treatment and control groups, the proportion of coarse sediments underlying the thalwag increased with increasing channel gradient and the two groups had very similar regression line slopes (Figure 4a; model r^2 = 19.4, model P value = 0.057, P value for differences in slopes of regression lines = 0.484, $n_{\text{treatment / control}}$ = 15/10). Conversely, point bar sediments at treatment sites had decreasing abundances of fines with increasing gradient, but control sites had increasing fines with increasing gradient (Figure 4b, model r^2 = 14.7, model P = 0.098, P for differences in slopes of regression lines = 0.049, $n_{\text{treatment / control}}$ = 15/10).

Macroinvertebrate abundance did not differ between treatment and control sites, but was affected by channel gradient and river segment (two-way ANCOVA #Inverts: co-variate Gradient w / 1 df, F ratio 3.92, P = 0.0604; Segment w / 2 df, F ratio = 2.95, P = 0.0735; Treatment or Control w/ 1 df, F ratio 0.01, P = 0.9409). Bessette Creek supported lower abundances than either the lower or middle Salmon River (mean [st error]: Bessette 296 [123], lower Salmon 552 [104], middle Salmon 466 [121]). Detecting differences in macroinvertebrate abundance between treatment and control sites with statistical power of 0.8 for a one tailed test at α = 0.1 would require sampling at 9,003 treatment sites and an equal number of control sites.

Characterizing site-to-site variability

NMS ordination required 41 iterations to produce a stable 3-dimensional solution (i.e., a solution with 3 axes) with final stress of 6.2 and instability of 0.005 (Figure 4). Stress is a measure of the suitability of the solution, as it indicates how well the solution reflects the structure of the original data set following the reduction in dimensionality. Instability is a measure of the magnitude of fluctuations in stress over the last 10 iterations of the ordination. The final stress and instability values of our solution are indicative of a stable solution (McCune and Grace 2002) with low risk for drawing false inference (Clarke 1993). The Monte Carlo test indicated our solution had lower stress than expected by chance (mean stress of Monte Carlo test: 15.0, test of difference between Monte Carlo and actual data P = 0.0476).

Cumulative variance explained in the solution was 95.7%, with axis 1 contributing 52.9%, axis 2–29.9%, and axis 3–12.9%. Axis 1 loaded with 5 parameters with correlations stronger than $\pm$ 0.5 along the axis: gradient, #Recruits, HabUnits, P TH Coarse, and Hgt IB. Axis 2 also had 5 variables load with correlations stronger than $\pm$ 0.5: D:W, P SA Slow, #Recruits, Cov T&S IB, P TH Coarse. Only gradient and P SA Slow loaded with correlation values >$\pm$ 0.5 on Axis 3. Treatment and control sites are not well

Table 3. Means ($\pm$ 1 standard deviation) of treatment (n = 16) and control (n = 11) sites, ANOVA P values for contrasts of treatment and control sites, and number of sites needed in both treatment and control groups to attain power of 0.8 (desired n). Means and standard deviations shown are for untransformed data, but statistical tests were done on transformed values. Abbreviations explained in Table 1.

Variable	Treatment	Control	*P* value	Desired *n* *
Cov T&S IB	0.604 (0.52)	0.582 (0.32)	0.926	4,504
#Recruits	6.690 (11.10)	5.550 (6.90)	0.826	902
Cov T OB	0.235 (0.31)	0.150 (0.27)	0.341	–
Cov S OB	0.585 (0.31)	0.147 (0.21)	<0.001	–

* Note: Power analysis to attain desired n assumes effect size equal to that observed in our present comparisons. Desired n was only determined for the two inside bank riparian parameter.

Figure 3. Predicted linear regression lines and associated data points for relationships between the proportion of thalwag sediments in the coarse size class and channel gradient (panel A) and the proportion of point bar sediments in the fine size class and channel gradient (panel B). A: Model r^2 = 19.4, model P value = 0.057, P for differences in slope = 0.4847. B: Model r^2 = 14.7, model P value = 0.098, P for differences in slopes = 0.049. The four circled data points had unusually large DFITS values but their exclusion did not affect tests of significance or interpretation and therefore they are included.

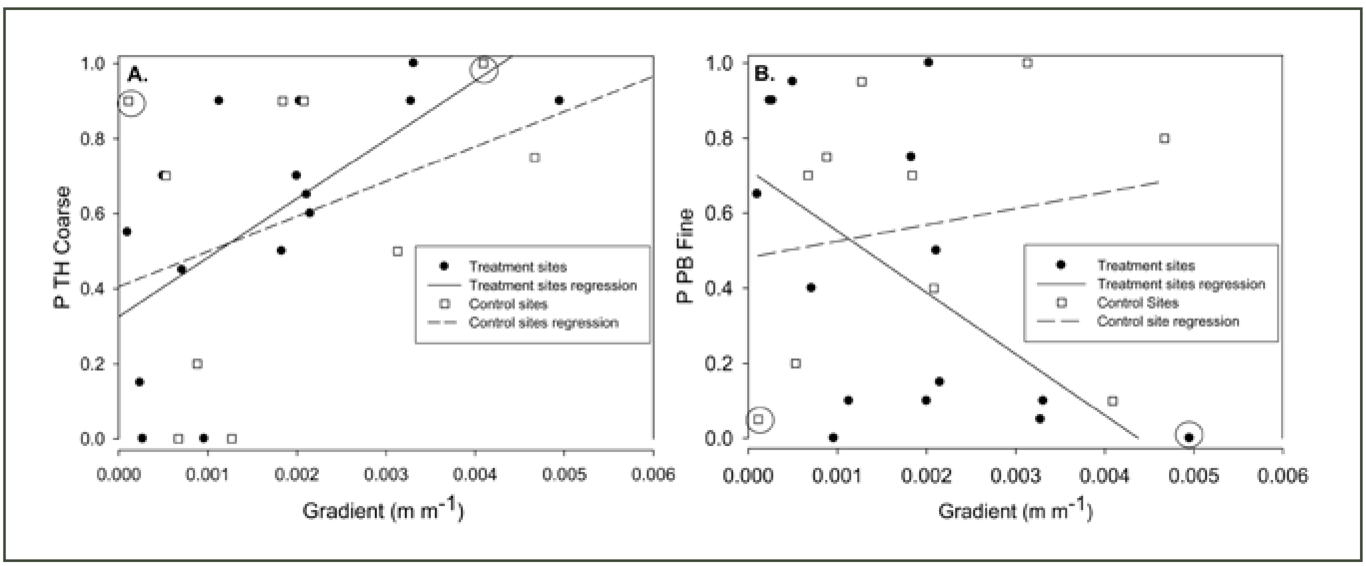
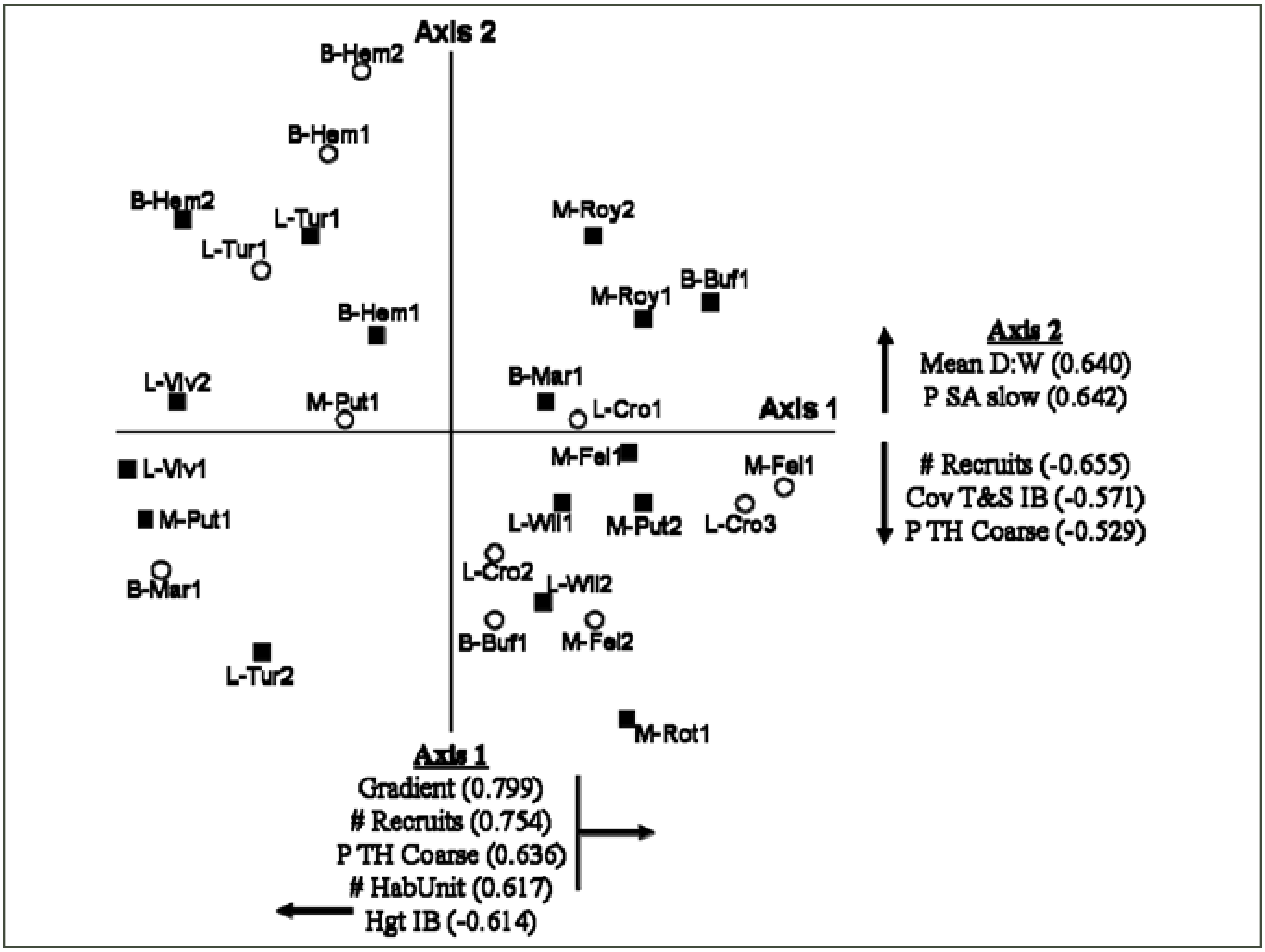


Figure 4. Results of a three dimensional non-metric multidimensional scaling ordination rotated to maximize correlation of gradient on axis 1. Axis 1 explains 52.9% of variance, axis 2 29.9% and axis 3 12.9%. Axes are labeled with variables that had correlation scores ≥ ±0.500 and arrows point in direction of increasing values. Solid squares are treatment sites (n = 16), open circles (n = 11) are control sites. The four intensive treatment sites are L-T5, M-T2, B-T4, and B-T3.



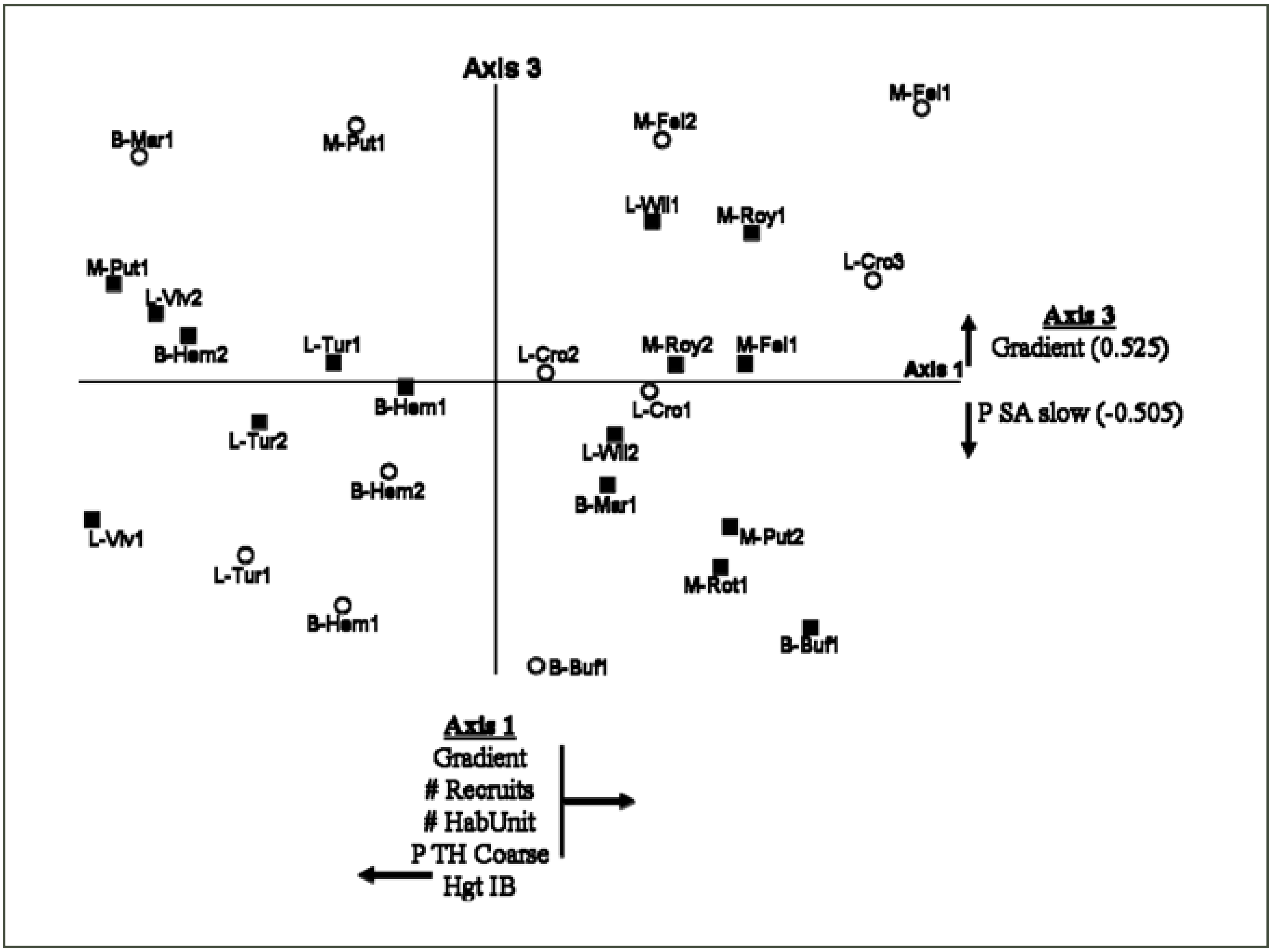
separated along any of the three axes, indicating the NMS ordination did not identify strong differences between treatment and control sites. Sites from the three river segments are similarly intermingled within the solution, indicating the three river segments had comparable habitat conditions. The 27 study sites are well distributed across all 3 axes, suggesting our solution is not influenced by a small number of unusually influential points (Figure 4).

DISCUSSION

We found limited evidence that stream restoration actions taken to stop riverbank erosion and improve salmonid production has substantially changed habitat condition at the scale of the river reach. Our multivariate tests are powerful tests of the hypothesis that bank restoration would yield measurable changes in habitat condition, as they can detect the cumulative effect of multiple small univariate differences. However, neither MANOVA nor NMS provided evidence that treatment sites substantively differed from control sites. However, treated sites did have narrower wetted widths, higher inside banks, and relatively more shrubs along outside banks, compared to unrestored control sites and these differences may be taken as indicators of the preliminary success of the restoration efforts and of processes that can lead to greater system recovery over time.

The narrower wetted widths present at treatment sites was a predicted outcome of restoration efforts and a goal of the restoration program in order to limit thermal heating of the water column. However, absence of pre-treatment and/or as-built data makes it impossible to determine if the observed width differences were a result of geomorphic adjustment to a new condition, such as sediment deposition along channel margins or channel downcutting, or simply a result of restoration actions which included dumping a large volume of material into the active channel. If narrowing was a result of bank accretion, then we can realistically expect the growth of the bank to continue until an equilibrium width is attained. If differences are a response to downcutting then undercutting and failure of treatments and a new wave of bank erosion may result. If, however, the width difference is simply equivalent to the width of the rock and wood added to the channel as part of the restoration, then there should be no expectation of future channel width

Figure 4 continued..



adjustment. Because channel width is one of the most reliable and indicative hydraulic variables for characterizing watershed condition (Andrews 1982; Woodsmith et al. 2005 and citations therein), it would be beneficial to our effectiveness evaluation to have better insight into the mechanisms responsible for stream width dynamics.

The inside bank at treatment sites had a higher mean elevation over the water surface compared to control sites, and this may be an indication that the floodplain is re-building in these locations. Changes in inside bank height was not a predicted response to treatment and the mechanism linking stabilization of the outside bank and height of the inside bank is unclear but may be related to bank treatment halting the prograding of the inside bend point bar into the active channel and thereby promoting stabilization of the point bar deposit and vertical accretion during subsequent high water events. Land surfaces with higher elevations are less prone to inundation and therefore less susceptible to disturbance during high discharge events, and reduced disturbance may promote increased survivorship of colonizing vegetation (Hupp and Osterkamp 1985; Friedman et al. 1995; Cooperman and Brewer 2005) and initiate a positive feedback between plant establishment, sediment accretion, and floodplain building resulting in a narrower channel (Cooperman and Brewer 2005). Treatment sites did not differ from control sites in number of natural recruits or total vegetative coverage on the inside banks but the former did have greater means and variances and there was a strong positive correlation between height of the inside bank and plant abundance (Cooperman, unpublished data).

Establishment of vegetation along outside banks of river bends is a well-documented technique for stabilizing riverbanks and potentially enhancing fish habitat. Given that all plantings done as part of the restoration program were at least four years old at the time of our assessment, our results provide an indication of good survival of the plantings. However, the primary plant used at these restoration sites, locally identified as "Pacific willow" (*Salix lasiandra*) though they are probably a cultivated hybrid, has neither a large central bole or a spreading canopy and they rarely exceed 3 m in height. These growth characteristics suggests these willows are unlikely to provide extensive shading to the stream (Roni et al. 2002) or serve as significant in-channel geomorphic features upon recruitment to the active channel.

The different relationships between channel gradient and proportion of fine sediments in point bar deposits between treatment and control sites was unexpected, especially since there were no differences between the groups in terms of particle sizes underlying the channel thalwag. A likely explanation is that treatment effectively diverted energy away from the outside bank of the river bend and onto the inside bank thereby winnowing away fine particles, and the greater the gradient the more stream power diverted. This raises the question of whether or not bank stabilization exports an effect, either upstream via channel downcutting or downstream via increased erosion, as has been documented for rip-rapped banks (Schmetterling et al. 2001). We had hoped to evaluate this potentiality, but the large site-to-site variability in the channel below study sites proved too large to disentangle from treatment effects (Cooperman, unpublished data). There was no relationship between sediment sizes in point bars and whether the next upstream meander bend was stable (with or without restoration treatment) or eroding (Cooperman et al. 2006).

Our macroinvertebrate results should be viewed cautiously owing to the samples being collected only from depositional environments (point bar deposits) where sediment particles are generally small. However, our treatment site point bars had relatively coarser sediments than those at control sites, and as coarser sediments often support more diverse and abundant invertebrate communities, it is reasonable to expect higher invertebrate abundances at treatment sites. The fact that we found no difference between treatment and control sites thus supports the conclusion that invertebrate biomass was not affected by restoration activities.

We did not test our data for spatial auto-correlation nor did we apply p-value corrections for multiple comparisons (i.e., Bonferronni adjustment). As such, our failure to find large geomorphic or ecological response to restoration can be viewed as conservative because had auto-correlation been present or post-hoc adjustments been applied, our statistical tests would have indicated even less differences between treatment and control sites than reported here (Hinch et al. 1993).

The NMS ordination illustrates the importance of pre-treatment data to effectiveness monitoring. Of the eight habitat measures used in the NMS analysis, four were strongly associated with gradient, the one parameter we measured that should be independent from presence-absence of bank restoration treatments. These four variables (HabUnits, P TH Coarse, Hgt IB, # Recruits) incorporate primary measures of reach-scale habitat diversity, sediment particle sizes, channel geomorphology, and riparian condition, and as such capture much of the ecologically relevant dynamics of interest. The strong correspondence within the NMS solution between gradient and these four variables suggests site-specific gradient is at least partly responsible for site-specific "ecology." Hence, site-to-site variability in our response variables associated with changes in local channel gradient confound our ability to determine the magnitude of change related to treatment efforts. Larson et al. (2001), in their discussion of the four types of variation that need be accounted for in monitoring programs (within-year at a site, independent year-to-year variation within a site, synchronous year-to-year variation among sites, and fundamental site-to-site variation), describe how failure to account for site-to-site variation, such as the gradient affect described here, can hinder trend detection. These authors suggest that multiple samplings at a site over time (i.e., a time-series approach comparable to before-after assessments) is the means to eliminate site-to-site variability.

Although certain statistical techniques can attempt to account for the influence of gradient on response variables, such as the use of regression residuals in a t-test comparing treatment to control sites, t-tests of residuals would still rely upon post-treatment between group comparisons and would be expected to have comparable statistical power to ANOVAs of the same data set. Also, using residuals may remove so much of the variability of interest as to overwhelm the ability to detect real between group differences. We tested regression residuals for treatment-control differences in depth-to-width ratio or height of the inside bank, the variables we measured most likely to be affected by local gradient. Results had *P*-values similar to those of our ANOVA tests and the t-tests suffered from comparably low statistical power (Cooperman, unpublished data). Only by holding gradient constant between treatment and control sites would we be able to fully disentangle the contribution of restoration from that of gradient.

The gradient effect exemplifies a weakness of the extensive post-treatment approach for effectiveness monitoring and it may be at least partly responsible for the low statistical power of our univariate comparisons. Our a posteriori power calculations indicate that for 7 of our 9 habitat variables, we would have needed over 100 sample sites (50 of each treatment and control) to attain a statistical power of 0.8, a standard value considered reasonable for ecological data (Peterman 1990; Steidl et al. 1997). Based on logistics of project planning, site selection, data collection within a limited field season, and the available budget, it was not possible for us to sample more than 16 treatment and 11 control sites. Even if time and money were more available, there were not many more sites that we could have examined which fit our project criteria. Thus, with "effect-size" changes of habitat variables on the orders we observed, it is not possible for us to definitively assess restoration effectiveness with the extensive post-treatment study design that we used. Our study would have greatly benefited from foresight to collect relatively inexpensive pre-restoration information (e.g., elevation cross sections, channel dimensions, rapid habitat unit, and riparian vegetation survey—total time ~ two hours for two trained people).

Our results illustrate an important lesson that should be heeded by agencies and groups wishing to conduct restoration activities and eventually assess their effectiveness. Only by using an experimental design capable of disentangling change caused by treatment from change caused by external factors and natural variability can definitive assessments of the affect of treatment result. Appropriately replicated and controlled before-after designs provide the suitable mechanism for restoration monitoring. Using a before-after approach (Green 1979), or its derivatives such as BACI, beyond BACI (Underwood 1991), and staircase design (Walters et al. 1988) not only eliminates the confounding influences of site-to-site and year-to-year variability on detecting response to treatment, it can provide greater statistical power with fewer replicates (Roni et al. 2005), thereby saving time and money. Although selection of appropriate control sites is always important regardless of use of post-treatment or BACI approaches (Roni et al. 2005), only the post-treatment design is wholly dependant upon the need to assure "control" sites are suitable matches to the treatment group. The ultimate choice of which design to use is a function of logistical and budget constraints, but even simple before-after comparisons offer greater potential to detect relevant trends than do after-the-fact assessments.

One question regarding before-after approaches that is unresolved is the number of times a site needs to be sampled, both pre- and post-treatment. We suspect the answer is study specific and depends upon investigator knowledge of the potential magnitude and rate of response to treatment that can be reasonably expected. Knowledge of the generation time of populations of interest and/or the return frequency of key geomorphic and/or disturbance events such as floods, drought, and fire seem reasonable starting points. In one case, a comprehensive BACI evaluation of stream restoration in Finland found evidence of response to restoration in habitat structure, benthic invertebrates, trout abundance, and ecosystem process based on a single pre-treatment and single post-treatment sampling (Muotka and Syrjänen 2007). Alternatively, Wooolsey et al. (2007) determined sampling frequency for assessing restoration effectiveness on the Thur River of Switzerland, on a variable by variable basis. For example, they assessed wetted width twice before restoration, three times after the first flood, and once after the second flood. For surface-hyporheic exchange, sampling occurred twice before treatment and once after the first flood; and, for lateral connectivity, sampling was once each, before and after.

The absence of wide ranging large differences between the treatment and control sites compared in this study does not mean the restoration program has not yielded benefits. The treatments have been highly effective at preventing further bank erosion, the proximal programmatic goal, and have successfully established riparian vegetation. Furthermore, although small sample size precluded statistical comparisons, the mean condition of the four "intensive treatment" sites suggests that more expensive and detailed site-specific actions may yield larger favorable changes in habitat quality (e.g., greater habitat diversity, deeper and narrower channels, less fine sediments). The four intensive sites had a lower mean channel gradient than the other treatment sites or the control sites, suggesting intensive treatment attained the greater results despite having less stream power available.

Palmer et al. (2005) suggested five criteria by which to judge if restoration is successful: (i) did a predefined guiding image exist for the effort (i.e., statement of purpose and goal), (ii) has the river's condition been improved, (iii) has a more self-sustaining system emerged, (iv) did construction cause lasting harm, and (v) were both pre- and post treatment assessments done. Our biggest limitation is the lack of pre-treatment data and because of this we have limited power to make the definitive conclusions we would like. Based on the remaining criteria, the bank stabilization program described herein has been a modest success. However, criteria of less tangible issues, such as increased social awareness of the linkage between land-use and ecological consequences and the evolution of a stewardship mentality, need also be considered. A companion study conducted at the same time as the effectiveness evaluation reported and covering several watersheds of the British Columbia southern interior found that as the amount of restoration work done on a stream increased, so to did landowner awareness and appreciation of habitat restoration. Further, the increased awareness of habitat issues was matched by a corresponding increase in adoption of ecologically more benign land-use behaviors such as more extensive use of livestock exclusion fencing to control riparian grazing (Branton et al. unpublished data). As such, in a sociological context, the stream restoration effort described herein appears to have been highly successful.

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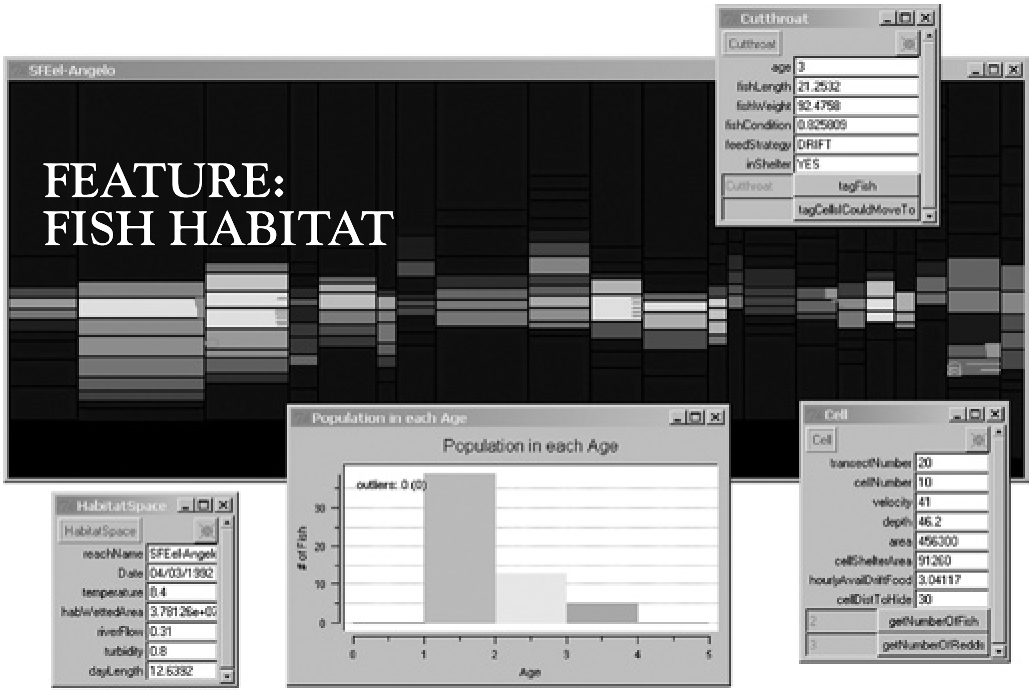
FEATURE: FISH HABITAT



inSTREAM's graphical interface illustrates the model's structure. The "HabitatSpace" window shows the daily inputs: flow, temperature, and turbidity. The main animation window is a top-down depiction of the habitat cells, with flow from right to left and cells shaded by depth (lighter is deeper). Fish appear as lines scaled by fish length, and redds as ovals. Depth and velocity of each cell vary daily with flow. Fish select the cell offering the best trade-off between growth and predation risk, a decision affected by temperature and turbidity as well as competition. Population status (illustrated in the age histogram) is determined by spawning and mortality of individuals. Users can open additional windows (via mouse clicks) to see habitat variables within a cell and the status of an individual fish; here: the fish tagged green and its current cell are shown. The tagged fish has low condition because it just spawned.

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Estimating Multi-Factor Cumulative Watershed Effects on Fish Populations with an Individual-Based Model

ABSTRACT: While the concept of cumulative effects is prominent in legislation governing environmental management, the ability to estimate cumulative effects remains limited. One reason for this limitation is that important natural resources such as fish populations may exhibit complex responses to changes in environmental conditions, particularly to alteration of multiple environmental factors. Individual-based models hold promise for estimating cumulative effects in these situations. We present an example application of an individual-based model of stream trout to the problem of estimating the cumulative effects of multiple environmental changes: elevated wet-season turbidity, elevated dry-season stream temperature, and reduced pool frequency. Each of these physical changes had multiple consequences for individual fish in the model, reflecting existing information. The simulations exhibited non-linear and non-multiplicative population responses to the multiple stressors. The results indicate the value of the individual-based approach for estimating cumulative effects and challenge the assumption that consequences for animal populations of increasing or multiple environmental changes are readily estimated from responses to modest changes in single factors.

Estimación de Efectos Multi-facoriales Acumulados de una Cuenca Hidrográfica en Poblaciones de Peces mediante Modelos Basados en el Individuo

RESUMEN: Si bien el concepto de efectos acumulados es importante en la legislación ambiental, su estimación aún es limitada. Una de las razones de dicha limitación es que recursos naturales de primera importancia como las poblaciones de peces pueden mostrar respuestas complejas ante los cambios ambientales, particularmente a la alteración de diversos factores. Los modelos basados en el individuo (MBI) son una alternativa promisoria para la estimación de efectos acumulados. En el presente trabajo se aplica un MBI a la trucha de río para estimar los efectos acumulados de distintos cambios ambientales: elevación de la turbidez durante la época húmeda, elevación de la temperatura del agua durante la época seca y un menor número de estanques disponibles. Cada uno de estos cambios físicos tuvo diversas respuestas en los organismos dentro del modelo, lo cual es un reflejo de la información existente. Ante los distintos factores forzantes, las simulaciones mostraron respuestas poblacionales no lineales y no multiplicativas. Los resultados ponen de manifiesto la utilidad de los MBI para estimar efectos acumulados y cuestionan la suposición de que las consecuencias que los cambios ambientales múltiples o crecientes tienen sobre las poblaciones animales, son fácilmente estimables a partir de respuestas a cambios individuales y de poca intensidad.

INTRODUCTION

The cumulative effects concept is fundamental to environmental policy and management, by virtue of its inclusion in key legislation such as the U.S. National Environmental Policy Act and the Canadian Environmental Assessment Act. However, many cumulative effects analyses focused on consequences for animal populations may not help achieve management goals, in part because of the inability to estimate biological responses to complex environmental changes (Duinker and Greig 2006). Most cumulative effects analyses must incorporate a variety of complicating aspects. For example, spatial variation can be important in cumulative effects: even within a watershed, the downstream transport of water, heat, and other watershed products may lead to accumulation of physical changes along stream networks (Li et al. 1994; Bolstad and Swank 1997). Temporal variation is often important: for example, significant cumulative effects may be undetectable until triggered by rare events such as extreme weather, and environmental changes separated in time may, as in the simulations we present, have cumulative impacts. Human activities often affect multiple environmental features important to the resource of interest. For example, changing the operation of a dam can alter not only flow but also water temperature and geomorphic processes downstream (Ligon et al. 1995), while timber harvest can affect temperature, turbidity, channel morphology, and possibly food production (Hartman et al. 1996). Finally, different kinds of human influences on the environment (e.g. habitat alteration and invasive species) commonly overlap (Allan 2004).

What tools are available to resource managers seeking to predict cumulative effects on key biological resources such as fish populations? Many of the models and tools fishery managers typically use for impact assessment are designed to evaluate static conditions and often best suited for evaluating effects of single factors. These approaches might be useful in cumulative effects analyses if one can assume that impacts are linear (e.g., a 50% decrease in preference-weighted habitat area reduces abundance by 50%) and that multiple impacts act independently. However, cumulative effects analysts whose qualitative reasoning suggests non-linear responses and interactions among factors have few tools to explore and expand their understanding.

Spatially explicit, individual-based models appear well-suited to support cumulative impact assessments. In these models, the consequences of environmental conditions on populations emerge from the effects of those conditions on individuals, which can often be effectively simulated. For example, Goss-Custard et al. (2006) used an individual-based model to identify critical thresholds of disturbance for wading birds by quantifying the energetic cost of disturbance and its consequences for over-winter mortality while incorporating spatial and temporal variation in food supply, the cost of thermoregulation, and other key factors controlling the energetics of individuals. Because they explicitly incorporate spatio-temporal variation in the environment and the mechanisms by which the environment affects individuals, the interacting effects of environmental variables on population dynamics can be directly estimated. Rose et al. (2000) described an individual-based model of striped bass (*Morone saxatilis*) in the Sacramento and San Joaquin river system that incorporates possible effects of diversion mortality, changes in prey composition that reduced food availability, and increased adult mortality on population dynamics. In that example, the combined effect of the three factors on the virtual population significantly exceeded the effect predicted by multiplying their separate effects.

One specific area where the individual-based approach may be useful to decision-makers is in the analysis of cumulative effects within watersheds, where human activities can influence a variety of physical factors and fish populations are often the foci of impact assessments (e.g., Hughes et al. 2006). Our objective in this study was to explore the use of an individual-based model in estimating cumulative watershed effects on stream trout, using realistic scenarios reflecting alteration of physical factors commonly influenced by human activities: water temperature, turbidity, and habitat structure.

METHODS

We used a spatially explicit, individual-based model (IBM) of cutthroat trout (*Oncorhynchus clarki*) in Little Jones Creek, northwest California, to explore the cumulative effects of physical changes commonly associated with human disturbance of forested watersheds. Little Jones Creek is a third-order tributary of the Middle Fork of the Smith River. The reach we simulated drains about 2,500 ha of forest, about 30% of which has been logged in the last 50 y. We used *inSTREAM*, an IBM we designed for assessment of changes in physical habitat on stream trout populations. This IBM has been described in detail elsewhere (Railsback and Harvey 2001; see also www.humboldt.edu/~ecomodel) and several examples of its use are available (Railsback and Harvey 2002; Railsback et al. 2003).

Brief description of the model

inSTREAM tracks the survival and growth of model fish within simulated stream reaches, using a one-day time step. Habitat is represented two-dimensionally as rectangular cells several square meters in size. In this example, we simulated one 372-m stream reach using habitat cells arranged along 42 transects. Variables representing the availability of hiding cover and velocity shelters for feeding are input for each cell and assumed constant over time. An external hydraulic model uses channel geometry and streamflow to determine the depth and velocity in each habitat cell. The model requires daily reach-scale values for streamflow, temperature, and turbidity.

Model trout may conduct four major processes in each time step: habitat selection, feeding and growth, mortality, and reproduction. Trout select the habitat cell they forage in considering habitat features and their own condition. Fish select the habitat cell, within a limited radius, that maximizes their probability of surviving (and, for juveniles, reaching reproductive size) over a future time horizon of 90 days (Railsback and Harvey 2002). This probability depends both on short-term risks such as predation, and on food intake (if food is inadequate, then starvation becomes a significant risk over the time horizon). The condition of individuals influences habitat selection by affecting starvation probability. Thus, a fish in poor condition might occupy a habitat cell with relatively high mortality risk if that cell also offers high food intake.

Feeding and growth (for which we adapted existing bioenergetics models) are affected by a variety of factors, including fish size, hydraulic conditions, turbidity, temperature, and competition. The model simulates two kinds of food: the concentration of drifting invertebrates and the production rate of benthic invertebrates are assumed constant on the reach scale.

Higher velocities carry more drifting food to the fish, but reduce the distance over which fish can see and capture food, and increase metabolic costs of swimming. Velocity shelters reduce swimming costs. Turbidity reduces the distance over which fish can capture drifting food, but does not affect benthic feeding. Fish compete for the food available in each habitat cell. This competition is assumed to be size-based: smaller fish only have access to food not consumed by the larger fish in their cell. Simulating competition for food has proven essential for reproducing a variety of realistic behaviors (Railsback and Harvey 2002; Railsback et al. 2002). Overall, growth is a function of food intake, metabolic activity, and temperature.

Mortality from specific sources is modeled by treating daily survival probability as a deterministic function of each fish's state and habitat; random numbers are then used each day to determine whether each fish survives. We model several different mortality risks, including predation, high temperature, starvation, and stranding. Predation by terrestrial animals is modeled separately from predation by fish because the risks of these two kinds of predation vary differently with habitat (e.g., water depth) and fish size. We assume elevated turbidity reduces predation risk from both terrestrial animals and fish.

Spawning is included in *inSTREAM* so that impacts of stressors on reproduction can be included and simulations much longer than one life span can be conducted. When environmental conditions are appropriate, model fish of adequate size in good condition can spawn. The nests of fish eggs are treated as individuals. Redds are vulnerable to several kinds of mortality: extremely high or low temperature, scouring in high flows, and dewatering in low flows. Egg development rate is controlled by temperature.

The model's software includes tools to thoroughly observe and understand the model. Graphical and file outputs are easily modified as needed to test and understand simulations. An animation window displays habitat conditions and fish locations. Thorough documentation and testing have been completed, including detailed user guides and a variety of rigorous testing procedures (Railsback and Harvey 2001). The software allows *inSTREAM* to be easily modified for new species and sites. Finally, an "experiment manager" automatically generates and executes model runs that compare scenarios and include replicate simulations.

The model requires several kinds of input. Parameter values are based on review of the literature and data on the process they represent (Railsback and Harvey 2001). Physical habitat input is site-specific. Using an established set of field procedures, a study site is selected and a grid of rectangular cells established. Variables representing feeding and hiding cover and spawning gravel are evaluated in the field. Depth and velocity in each cell is measured at several flows to calibrate a hydraulic model used to predict depth and velocity in each cell at each simulated flow rate. (We used PHABSIM hydraulic models for this study; alternative versions of *inSTREAM* interpolate depth and velocity directly from field measurements, or use two-dimensional hydrodynamic models.)

Time-series input includes daily values for stream flow, temperature, and turbidity. Here, we use 15-year records (October 1990–September 2005). Baseline conditions for Little Jones Creek represent actual measurements for 1998–2005. We estimated the discharge record for 1990–1998 using a strong linear relation between Little Jones Creek streamflow and Smith River streamflow at a USGS gage downstream (R^2 = 0.96 with a 1-h lag between sites). Also for the simulation period preceding direct measurements, we estimated temperature by establishing monthly relations between Little Jones Creek water temperature and the air temperature at a weather station in Cave Junction, Oregon. The coefficients of determination (R^2) ranged 0.5–0.8 for these regressions. Finally, to estimate turbidity for 1990–1998, we used a linear relationship (R^2 = 0.68) between turbidity and discharge at the study site. We chose not to use power functions to describe the relationship between turbidity and stream discharge, because of their tendency to overestimate turbidity at low discharge.

We calibrated the model using parameters representing especially important but uncertain environmental processes: food availability and predation risk. Five variables (representing drift food concentration, drift regeneration rate, benthic food production rate, maximum aquatic predator risk, and maximum terrestrial predator risk) each dominate one model output (adult growth, adult density, juvenile growth, juvenile mortality, and adult mortality). Typically the only calibration *inSTREAM* needs is adjustment of these five variables to produce realistic values for the related five outputs. We do not consider the IBM's sensitivity to these uncertain variables a flaw: it is a reflection of the reality that food availability and predation risk are very important drivers of fish populations yet they are inherently difficult to estimate. Note that while these variables are constant at the reach scale (e.g., the model assumes a constant concentration [mg of prey/m^3 of water] of drift food throughout a reach), variation in physical conditions among habitat cells produces strong differences in food availability and predation risk within simulated reaches.

In general, individual-based models predict the effects of stressors by simulating: (1) how the stressors affect an individual; (2) how individuals respond behaviorally to the stressor (here, primarily by selecting different habitat), considering their current habitat and competitive conditions; and (3) the consequent survival, growth, and reproduction of individuals, which when summarized over time yields an estimate of the population-level response (Grimm and Railsback 2005). Consequently, the effects of stressors on simulated populations can be complex and unexpected. Consider the effects of a stressor (e.g., sublethal but high temperature) that reduces individual growth. An individual exposed to the stressor would experience reduced growth, causing it to adapt by shifting habitat to sites with higher growth but higher mortality risk. Thus, the population-level result may be lower abundance (because more fish use riskier habitat and get eaten) as well as (or instead of) a decrease in average body size. Alternatively, mean body size could increase if a severe reduction in abundance reduced intraspecific competition to the point that the direct effect of sublethal high temperature on individuals was more than offset.

Models as complex as *inSTREAM* naturally raise concerns about uncertainty, especially the extent to which conclusions from the model could be artifacts of detailed assumptions and parameter values. While we do not analyze uncertainty in the results presented here, we have taken a number of steps to deal with the issue. The assumptions and parameters in the model are based to the extent possible on published information and thoroughly documented. A study of parameter uncertainty (Cunningham 2007) found the model highly sensitive to only a few parameters other than those either estimated via calibration or with values well-established in the literature, and no evidence of unpredictable or strong "error propagation." The same study also found a relative comparison of scenarios (the approach used here) robust to parameter uncertainty.

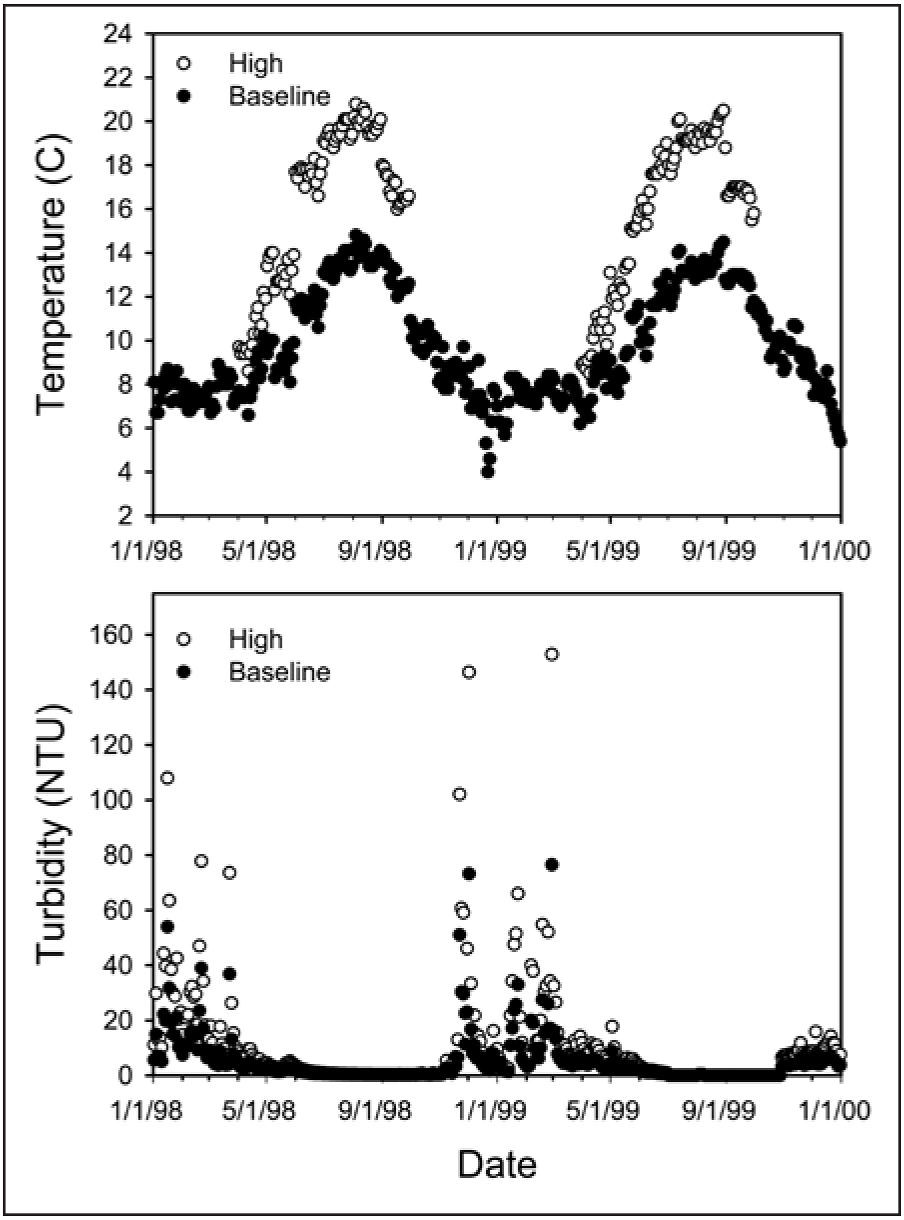
Simulation of cumulative watershed effects

Here we explore the cumulative effects of three physical changes commonly associated with human activities in temperate forested watersheds: increased dry season temperature (Moore et al. 2005), increased wet season turbidity (Gomi et al. 2005), and reduced pool frequency (McIntosh et al. 2000). For each of these changes, we establish three regimes: baseline (current conditions at Little Jones Creek), moderate (halfway between the baseline and high regimes), and high (a major change that is still well within observed conditions in the region). The high temperature regime (Figure 1) was derived from our unpublished data for disturbed streams in northwestern California. We created alternative turbidity regimes by increasing the slope of the turbidity—discharge relationship at Little Jones Creek by 50 and 100%. These alternatives are modest in comparison to turbidity—discharge relationships for some streams in the region of similar size, one of which has a turbidity—discharge relationship with a slope 720% higher than Little Jones Creek. While the modified temperature regimes only differed from the baseline regime during five months in the dry season, the elevated turbidity regimes have meaningful differences from baseline only during winter and occasionally spring stormflows (Figure 1). Finally, we reduced pool frequency in the simulated reach by 50% (the high impact regime) and 25%, replacing habitat cells along transects through pools with cells from adjacent run habitat. When cross-classified, these alternative regimes (including baseline conditions for all three factors) yielded 27 different scenarios.

Each physical alteration included in the simulations has multiple consequences for fish in the model. Temperature affects bioenergetics (the relation between food intake and growth) and some mortality risks. The model also simulates the potentially important effects of temperature on the duration and success of egg incubation, which can affect population size. Moderate turbidity can have both negative and positive effects on fish. By reducing visibility, turbidity reduces the amount of food fish can capture by reducing their reactive distances to drifting prey but also reduces the risk of fish being captured by some predators. Both of these mechanisms are represented in *inSTREAM*. Availability of pool habitat influences the survival and growth of fish in the model through effects of water velocity and depth on the energetics of feeding and on predation risk.

We contrasted simulation results by quantifying median population biomass over the last 10 years of the 15-year simulation period, for 10 replicates of each of the 27 scenarios. We also quantified population persistence as the percent of replicates in which the population did not go extinct. We increased the level of replication when persistence varied substantially. Variation among replicates in this case indicates only the influence of stochastic processes affecting individual fish. The main stochastic process in the model is mortality: the daily probability of survival for each trout is a deterministic function of its state and its habitat, but whether the trout actually lives or dies each day is a stochastic event. Given the relatively small length of stream simulated here, persistence is basically a measure of the population declining to a very low abundance, at which the probability of chance "extinction" is magnified. To explore non-linearities and interactions among factors, we compared results for population biomass and persistence to the results predicted by assuming effects are linear and multiplicative. For example, the predicted effect of the high temperature regime, assuming linearity, was twice the effect of the intermediate regime, while the predicted combined effect of high temperature and high turbidity regime was the product of their separate effects.

Figure 1. A two-year example of the input data for different temperature and turbidity regimes used in simulations of a resident trout population using an individual-based model. While the model uses a daily time step, for clarity the graph shows data from every other day.



RESULTS

Simulations of cutthroat trout abundance under baseline conditions at Little Jones Creek yielded a relatively stable population, similar to our population estimates for the site over the last eight years (B. C. Harvey, unpublished data). This result in part simply reflects successful calibration, but the multi-year results suggest the model also captured the influence of annual variation in patterns of streamflow, temperature, and turbidity. The moderate temperature, turbidity, and pool reduction alterations separately yielded modest biomass reductions (25% or less) compared to the baseline scenario, but the severe alteration regimes and the combined effects of multiple factors produced more dramatic effects (Figure 2). Both non-linear and non-multiplicative effects emerged from the simulations.

All noteworthy departures from predictions assuming linear, multiplicative effects were negative. The IBM predicted that doubling the change in each single factor more than doubled the impact on population biomass. The effects of multiple factors on mean biomass were often stronger than predicted by assuming each factor acts independently. Treatment combinations including increases in both water temperature and turbidity yielded particularly large departures from expected results assuming multiplicative effects (Figure 3), even though the within-year timing of these alterations did not overlap. Also, when combined with decreased pool frequency, the interaction of temperature and turbidity effects occurred even when both were only moderately altered. Departure from multiplicative results over the last 10 years of the simulations was greatest under the most challenging environmental conditions that yielded very low mean biomass, suggesting that differences in population persistence influenced these results.

The strongly non-multiplicative results for population persistence (Figure 4) make clear their influence on the biomass results. None of the simulated populations went extinct in scenarios with just one factor altered to either a moderate or high level. Thus, the assumption of multiplicative effects would predict 100% persistence in all scenarios. However, the fish in simulations with multiple factors altered commonly did not persist over the 15-year simulation period. The most severe scenario, combining high temperature, high turbidity, and 50% pool reduction, yielded no persistence.

DISCUSSION

The IBM simulations presented here yielded frequent non-linear and non-multiplicative outcomes, particularly for population persistence. These results appear germane for decision-makers facing questions about the consequences of additional human activities in previously altered systems: the effects of additional alterations may often be greater than predicted by the system's response to prior changes. While providing a general illustration of the potential complexity of cumulative effects, the variable outcomes in these simulations also suggest the usefulness of specific analyses for particular populations and scenarios.

Why the preponderance of non-linear, non-multiplicative effects, particularly for population persistence? The IBM suggests two reasons. First, many of its relationships for how physical factors affect individuals are non-linear. Examples include the relationships between metabolism and temperature and between drift feeding success and turbidity. Because population-level responses to altered physical conditions arise from such individual-level relations, there is good reason to expect these responses will be non-linear. Second, non-linear relations between physical factors and the fitness of individuals are likely to produce complex patterns in the amount of useful living space when environmental conditions create increasing physiological challenges. Deteriorating environmental conditions can eventually reach thresholds: populations crash when conditions degrade to the point that little or no habitat allows individuals to survive, grow, and accumulate energy for reproduction. Consequently, predictions based on small perturbations become less reliable as the level of perturbation increases.

The simulations presented also suggest that process linkages between environmental changes and individual fitness should be considered when estimating cumulative effects. Different kinds of environmental change may be more likely to produce non-multiplicative results if they produce similar consequences for individuals, even if those consequences are separated in time. In this case, both elevated temperature and elevated turbidity had energetic costs to individuals, although these effects generally did not overlap within years. These factors combined to produce the most strongly non-multiplicative results.

To be useful, even complex models like *inSTREAM* must simplify reality. Therefore, uncertainties concerning the effects of environmental variables on individuals require

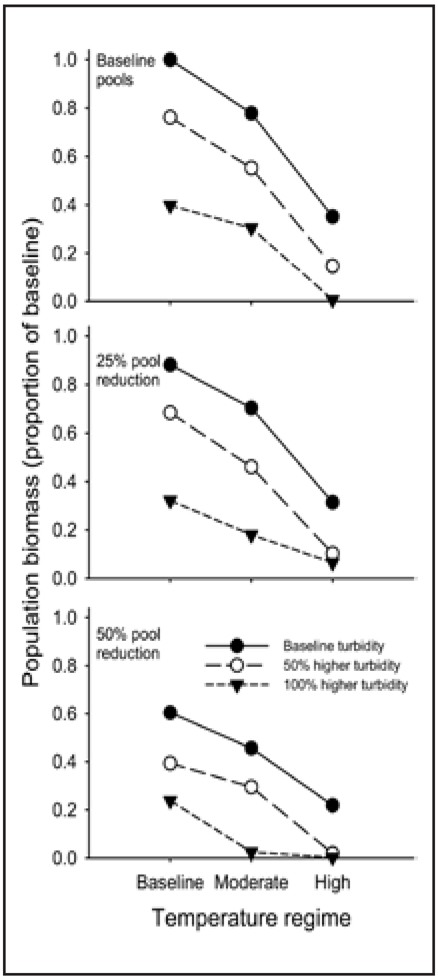
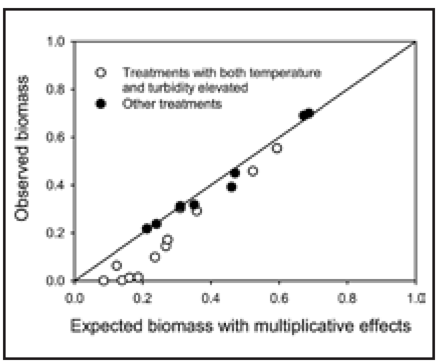
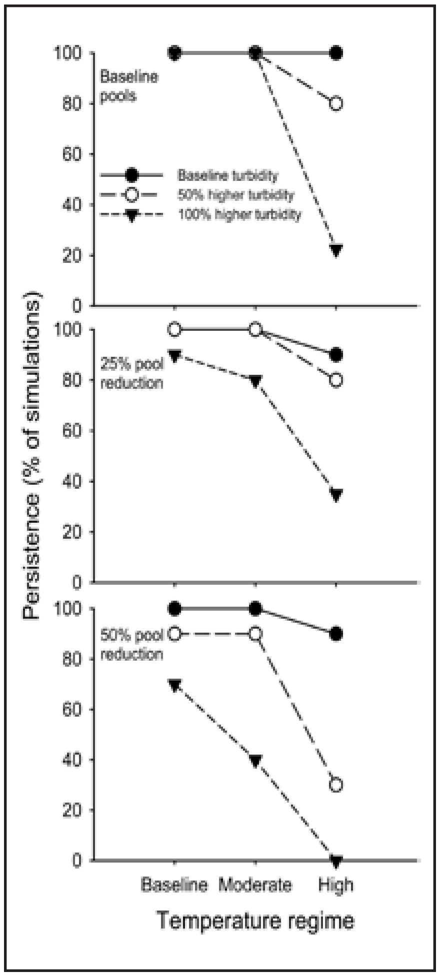


Figure 2. (left) Results of IBM simulations of a resident trout population under 27 different scenarios varying in turbidity, temperature, and pool frequency. The response variable reflects median biomass for 10 replicate runs of each treatment combination over the last 10 years of a 15-year simulation period, expressed as a proportion of the median biomass under baseline conditions.

Figure 3. (below) Comparison of IBM results for median biomass versus those expected given multiplicative effects of individual factors. Twenty treatment combinations included two or three altered environmental factors, thus allowing computation of an expected multiplicative effect. The graph distinguishes treatments in which both temperature and turbidity were altered from other treatments involving two altered factors.

Figure 4. (right) Results of simulations of a resident trout population showing the proportion of populations persisting for 15 years under 27 different treatment combinations. Sample size ranged 10–40. We completed more replicate runs for scenarios with intermediate levels of persistence.





caution in interpreting the quantitative results of simulations such as these. For example, we assumed trout cannot reproduce unless their weight is above 95% of "normal" for their weight, a substantial simplification of reproductive energetics. However, the potential significance of such uncertainties is readily addressed with sensitivity analyses. For example, completely eliminating the weight threshold for reproduction still yielded only 30% persistence under the most challenging scenario; our conclusion that persistence is much less than expected from assuming impacts are multiplicative therefore appears robust to how reproductive energetics are modeled.

Another key question in modeling efforts like this one is, have important linkages between the organism and altered environmental conditions been excluded? For example, elevated temperature can be associated with greater secondary production of lotic insects (Morin and Dumont 1994), which might alleviate to some extent the increased energetic demand of higher temperatures on fish. We did not include this possibility in the simulations. For a second example, *inSTREAM* includes the well-quantified effects of turbidity on the ability of trout to feed on drift (Sweka and Hartman 2001), but ignores the possibility that benthic prey may be more available during high-flow events that elevate turbidity.

The reach-scale model used here does not address larger spatial scales that must be incorporated in cumulative effects analyses focused on entire animal populations or metapopulations. Clearly, persistence is affected by a variety of landscape-scale processes (Dunning et al. 1992). Larger scales can be addressed with *inSTREAM*: it can simulate reaches of any size, and even networks of reaches connected by movement corridors. However, the usefulness of complex IBMs is limited by requirements for input and computing power. Given such limitations, another possible approach to cumulative effects analyses would use reach-scale individual-based models to inform less detailed models formulated for larger scales (e.g., Rieman et al. 2001).

From our perspective, the uncertainties and limitations of this approach are offset by the fact that explicit simulation of interactions between individuals and physical habitat that varies realistically in space and time can offer unique insights into the responses of populations to environmental changes, and the mechanisms responsible. Given the potential for alteration of multiple environmental factors to influence animal populations in complex ways, it may often be useful to apply an approach that explicitly incorporates such complexities.

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Aug 12-18—**30th Congress of the International Association of Theoretical and Applied Limnology: Redefining Theoretical and Applied Limnology in the 21st Century,** Montreal, Canada. See www.sil2007.org.

Aug 15-16—**European Aquaculture Society: Aqua Nor Forum 2007,** Trondheim, Norway. See www.easonline.org/home/en/default.as.

Aug 22-24—***Salvelinus confluentus* Curiosity Society Annual Meeting,** Perkins Lake, ID. Contact Dan Kenney, dkenney@fs.fed.us, 208/622-0094.

Aug 24-26—**Zhanjiang 2007 International Prawn Exhibition; International Aquatic Products, Reservation, and Transportation Equipment Expo; and International Aquatic Breeding, Processing Technology, and Equipment Expo,** Guangdong, China. See www.southfish.com.cn.

Sep 2-6—**American Fisheries Society 137th Annual Meeting,** San Francisco, CA. See www.fisheries.org/sf/.

Sep 11-15—**Fish Stock Assessment Methods for Lakes and Reservoirs Conference: Towards the True Picture of Fish Stock,** Ceske Budejovice, Czech Republic. See www.fsamlr2007.czweb.org.

Sep 17-21—**International Council for the Exploration of the Sea,** Helsinki, Finland. See www.ices.dk.

Sep 18-21—**International Conference on Freshwater Habitat Management for Salmonid Fisheries,** University of Southampton, UK. See www.salmonidhabitat.co. Contact Lynn Field, admin@salmonidhabitat.com.

Oct 8-11—**Second International Symposium on Tagging and Tracking of Marine Fish with Electronic Devices,** San Sebastian, Guipuzcoa, Pais Vasco, Spain. See http://unh.edu/taggingsymposium/.

Oct 9-10—**Seattle-Bioneers Conference 2007,** Seattle, Washington. See www.nwetc.org.

Oct 9-12—**International Symposium: Wild Trout IX,** West Yellowstone, MT. www.wildtroutsymposium.com/. Contact Dirk Miller, Dirk.Miller@wgf.state.wy.us, 307/777-4556.

Oct 15-17—**Third International Sustainable Marine Fish Culture Conference,** Fort Pierce, FL. See www.sustainableaquaculture.org. Contact Amber Shawl, ashawl@hboi.edu, 772/465-2400 x578.

Oct 15-18—**Institute of Fisheries Management Conference,** Westport, Co. Mayo, Ireland. Contact Stephen Axford, stephen.axford@environment-agency.gov.uk.

Oct 18-20—**Recirculating Aquaculture Systems: Principles of Design and Operation,** Fort Pierce, FL. See www.aquaculture-online.org. Contact Amber Shawl, ashawl@hboi.edu, 772/465-2400 x578.

Oct 21-24—**Southeastern Association of Fish and Wildlife Agencies Annual Meeting,** Charleston, WV. See www.seafwa2007.org.

Oct 23-24—**Sixth Practical Short Course: Aquafeed EURO-ASIA 2007: Aquaculture Feed Extrusion, Nutrition, and Feed Management,** Istanbul, Turkey. See www.membraneworld.com/aquafeed euroasia.htm. Contact Ignace Debruyne, aquafeed@scarlet.be, +32 (0)51 31 12 74.

Oct 23-25—**Fundamental Contaminant Chemistry: A Review of Chemistry Principles Essential for Understanding Contaminant Behavior in the Environment,** Honolulu, Hawaii. See www.nwetc.org/chem-403b_10-07_honolulu.htm. Contact Kristine Robinson, krobson@nwetc.org.

Oct 24-25—**Contaminant Chemistry and Transport in Soil and Groundwater: An Overview of Petroleum, Chlorinated Hyddrocarbon, and Metal Behavior in the Environment,** Honolulu, Hawaii. See www.nwetc.org/chem-403a_10-07_honolulu.htm. Contact Kristine Robinson, krobson@nwetc.org.

Oct 24-27—**Aquaculture Europe: European Aquaculture Society and Eurasia Trade Fairs,** Istanbul,Turkey. See www.easonline.org/agenda/en/AquaEuro2007/Aqua200.asp.

COLUMN: GUEST DIRECTOR'S LINE

2008 Tagging Symposium Steering Committee

See www.fisheries.org/units/tag2008 for more information.

Symposium on Fish Marking and Tagging

The American Fisheries Society, the Australian Society for Fish Biology, and the New Zealand Marine Sciences Society are pleased to announce "Advances in Tagging and Marking Technology for Fisheries Management and Research," to be held in Auckland, New Zealand, 24–28 February 2008.

Measurement is the key component in most investigations of fish and shellfish. The ability to identify individual and groups of fishes, as well as their habits, movements, and mortality, is crucial to effective fisheries science. The methods used must be appropriate, accurate, and repeatable. While uncertainty is an integral part of dealing with biological systems, as scientists it is crucial that we use methods that minimize uncertainty in order to improve the conservation and sustainability of fisheries and aquatic resources.

In June 1988, over 400 fisheries and aquatic scientists gathered in Seattle, Washington, for the "International Symposium and Educational Workshop on Fish-Marking Techniques." This landmark event included presentations on virtually every fish tagging method in use at that time. The ultimate product was the 1990 publication *Fish Marking Techniques,* American Fisheries Society Symposium 7, arguably one of the most influential fisheries publications in decades.

In the nearly 20 years since that symposium, the world of fisheries science has changed dramatically; the technologies and analytical procedures available for marking and monitoring fisheries have evolved as well. Fish marking technologies on the cutting edge two decades ago are now commonplace, and new technologies are developed yearly. Clearly, the time has come to bring together again global expertise on fish tagging techniques and data analysis.

The sessions for "Advances in Tagging and Marking Technology for Fisheries Management and Research" will include satellite tags, archival tags, acoustic tags and arrays, radio telemetry, new methods utilizing traditional internal and external tags, chemical and genetic marks, various integrated approaches, and data analysis techniques. It is the hope that discussions held at this symposium will be the impetus for even greater advances in tagging for fisheries science. The proceedings of this new symposium will provide the next step beyond *Fish Marking Techniques,* the proceedings from the Australian Society for Fish Biology tagging workshop in 1988, and the "Workshop on Fish Movement and Migration" in 1999, into this century's methods, technologies, advances, and challenges.

The Program Committee continues to compile an impressive roster of keynote speakers for the conference, including Barbara Block, John Sibert, Julian Metcalfe, Steve Campana, Michelle Heupel, and more. The call for papers and posters is up on the web site, so your abstract submission is welcome. Whether you present or attend, don't miss this opportunity to participate.

Our venue will be the University of Auckland. The trade show will be directed specifically toward companies dealing in the fields of marking and tagging. Other elements include a *Powhiri,* which is a ceremony of welcome extended by Maori, the indigenous people of New Zealand. The various elements of the *Powhiri* serve to ward off evil spirits and unite both visitor and host in an environment of friendship and peace. Two socials and

a conference dinner will provide ample opportunity to mingle and exchange information with colleagues new and old.

The conference also offers what, for many, may be a once in a lifetime opportunity to visit beautiful New Zealand. With vast open spaces filled with stunning rugged landscapes, gorgeous beaches, geothermal and volcanic activity, a temperate climate, and fascinating animal and plant life, it is no surprise that New Zealand's pure natural environment is so attractive to visitors from other countries. Another great advantage of New Zealand is there are many different landscapes, environments, and ecosystems so close to each other. February is late summer in New Zealand, with long days and mild nights, perfect for any outdoor activity. Being in such close proximity to Australia, our colleagues from there suggest a side trip as well. Everything from the Great Barrier Reef to the Outback could only add to the experience.

For more information, visit our website at www.fisheries.org/units/tag2008, or feel free to contact brad.parsons@dnr.state.mn.us. To see what awaits you in the southern hemisphere, you can visit the official tourism websites for New Zealand and Australia at www.newzealand.com/travel and www.australia.com

COLUMN: PRESIDENT'S HOOK

Continued from page 264

Information Technology Workshop

Tuesday, September 4, 5:30–7:30 p.m., Marriott Hotel, Golden Gate Room C

One additional topic we discussed in Bethesda that is important to the general membership was an issue brought up at the Atlanta mid-year Governing Board meeting—electronic publication of *Fisheries*. In Atlanta the Governing Board passed a recommendation that I establish a committee to look into electronic publication of *Fisheries*, which could then be delivered electronically and/or in print, subject to the wishes of AFS members. Making *Fisheries* electronically available to members is not a simple change and has financial implications for AFS. Production costs of *Fisheries* are primarily supported by advertising in the journal, not by publication page charges or library subscriptions. This journal is currently marketed to advertisers as a print journal with certain expectations of shelf life and access based on hard-copy print production. The transition of *Fisheries* to an elective-electronic journal may change traditional advertising revenues and force an increase in the membership dues structure to compensate for lost revenues. To avoid this potential funding gap for the journal, I asked Gus to task the *Fisheries* editorial staff to look into new electronic advertisement opportunities and to ask for information from current advertisement contracts about options for a transition to an electronic format for this journal. There is also the question of distribution once we have decided on a new format—should electronic *Fisheries* remain open to the membership only or become open access? Electronic distribution allows broader sharing opportunities but at a loss of control in the distribution structure. AFS staff will report on their findings for electronic publication of *Fisheries* to the new ESAB, with a follow-up report and recommendation from the ESAB to the Governing Board at the mid-year meeting in 2008.

I truly expect a sea change in IT services at AFS over the next year. The leadership at AFS and your elected officers are dedicated to these changes and to new electronic services opportunities for the membership. The AFS executive director and staff are committed to the strategy outlined above. Society leadership will determine the appropriate resources necessary to make this vision a reality. I'm sure follow up will be efficient and effective, but we remain open to comments and suggestions from the full membership. Important issues of outsourcing and/or in-house development of electronic services, appropriate software applications and updates, and integrated electronic database management at AFS need to be discussed and worked out in a cost-effective IT strategy. This is why an active and comprehensive Electronic Services Advisory Board is required at this time. We need a new and effective programmatic link for IT at AFS that marries the needs of the membership with the tools available in a dynamic and ever-changing electronic information environment.

Proposed Constitutional Amendment

The Governing Board approved renaming and expanding the mandate of the Web Editorial Advisory Board at their March 2007 meeting in Atlanta, Georgia. To be adopted, a proposed constitutional amendment must be published at least 30 days in advance and approved by at least a 2/3 vote of the general members present at the AFS Annual Business Meeting in San Francisco, California, on 4 September 2007. Any related changes to the Procedures Manual may be approved by the Governing Board.

CONSTITUTION IX. Standing Committees

Current:

2.Z. WEB EDITORIAL ADVISORY BOARD provides oversight on the content and structure of the AFS WEB site. Its goal is to maintain high standards of technical content and presentation on the web site according to policies set forth in the current Strategic Plan of the American Fisheries Society and current plan for the AFS Web Site.

Proposed:

2.Z. ELECTRONIC SERVICES ADVISORY BOARD provides oversight and coordination for electronic membership services, including those pertaining to AFS communications, publications, meetings, unit functions, and the content and structure of the AFS web site. Its goal is to maintain high standards of technical content and presentation, ease and continuity of membership access, and review of new electronic tools for membership services according to policies set forth in the current Strategic Plan of the Society.

OBITUARY: DIRECTOR OF THE COLUMBIA RIVER RESEARCH LABORATORY

James H. Petersen

James H. Petersen, director of the U.S. Geological Survey's Columbia River Research Laboratory (CRRL), in Cook, Washington, passed away suddenly at work on 22 March 2007, at the age of 53.

Petersen was born in Emmett, Idaho, and grew up in McCall, Idaho, where he attended high school and graduated as the salutatorian. He then attended Boise State University, graduating in four years. Shortly thereafter he received a Rotary scholarship to study marine ecology in the Great Barrier Reef off the coast of Australia. Upon return to the United States in the late 1970s, he entered the University of Oregon where he continued studies in marine ecology, earning his Ph.D. in 1984. From 1984–1987 he worked for the Los Angeles County Natural History Museum as a marine ecologist. While in southern California, Petersen conducted research on the effects of nuclear power plant cooling water effluent on kelp bed ecology. In 1988 he accepted a position as a research fishery biologist at the CRRL, operated by the U.S. Fish and Wildlife Service at that time (the laboratory transferred to the U.S. Geological Survey in 1997). For over a decade, Petersen examined numerous aspects of predation as a factor limiting the survival of juvenile salmonids in the Columbia River Basin. That research included examining the behavior of predators and their prey, developing bioenergetics and individual-based models of predation, and projecting changes in predation in a "normalized" Columbia River. In 1994, Petersen was a key driver in developing the stream ecology program, which has become a mainstay of CRRL. More recently, he was working on a wide range of topics, including the impacts of altered water temperature on native fishes in the Columbia and Colorado rivers, the influence of invasive species on survival of Pacific salmon, and the effects of altered water quantity and quality on the thermal endemic aquatic community in the Muddy River near Las Vegas, Nevada. Petersen's contributions to aquatic sciences are chronicled in over 25 peer-reviewed publications, numerous technical reports to the fisheries agencies that supported his research, and his appointment to adjunct faculty at the University of Washington and the University of Idaho.

During his all-too-short tenure at the CRRL, Petersen received numerous performance awards, culminating in a prestigious Fulbright Fellowship to Jamaica where he taught bioenergetics modeling. He was a member of the American Fisheries Society and the Ecological Society of America. Petersen was a scientist of international renown who collaborated with other researchers throughout the United States and abroad. He was a mentor to many young fishery biologists working in the region, an effective leader with quiet grace and respect for all, and a close and dear friend to many. As director of CRRL, Petersen oversaw the work of as many as 150 employees and helped to position the lab as one of the preeminent fisheries research facilities in the Pacific Northwest. His staff looked up to him as someone who led through example, kind words, and a genuine concern for the research and the people conducting it.

He married Dena Gadomski in Long Beach, California, in 1989, and she moved to Cook, Washington, to join her husband sharing love, life, and fisheries research. One of their passions was to travel, and they shared many exciting trips together to various far away locales including Scotland, Italy, Central America, the Cook Islands, Poland, Belgium, and France. His hobbies were birding, photography, and woodworking, at which he excelled in his fine attention to detail.

Contributions in Petersen's name may be made to the Audubon Society or The Nature Conservancy.

—Thomas Poe and Alec Maule

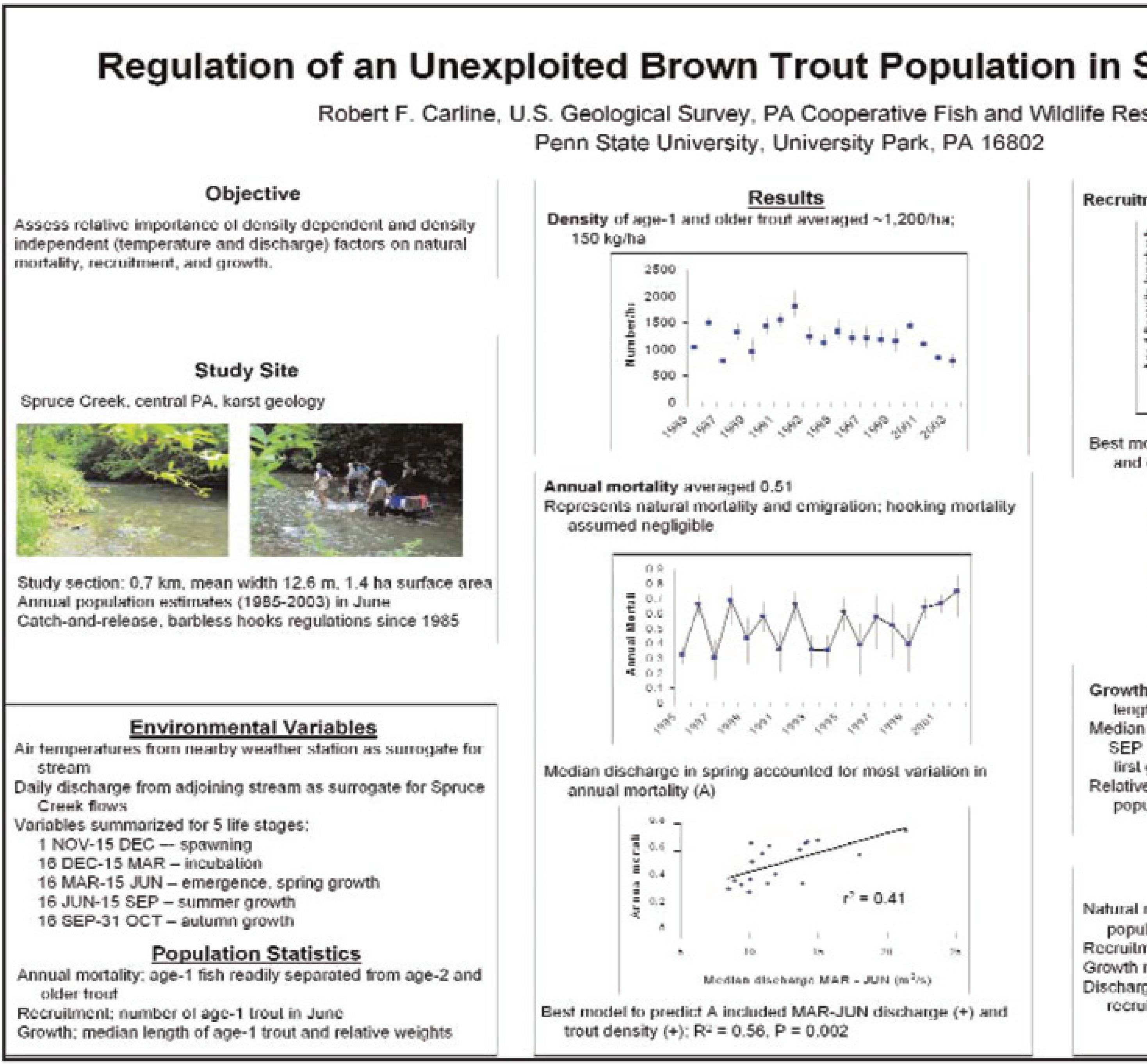
Guidelines for Designing Posters

Robert Carline

Carline recently retired from the U.S. Geological Survey, Pennsylvania Cooperative Fish and Wildlife Research Unit, Penn State University, University Park. He can be contacted at f7u@psu.edu.

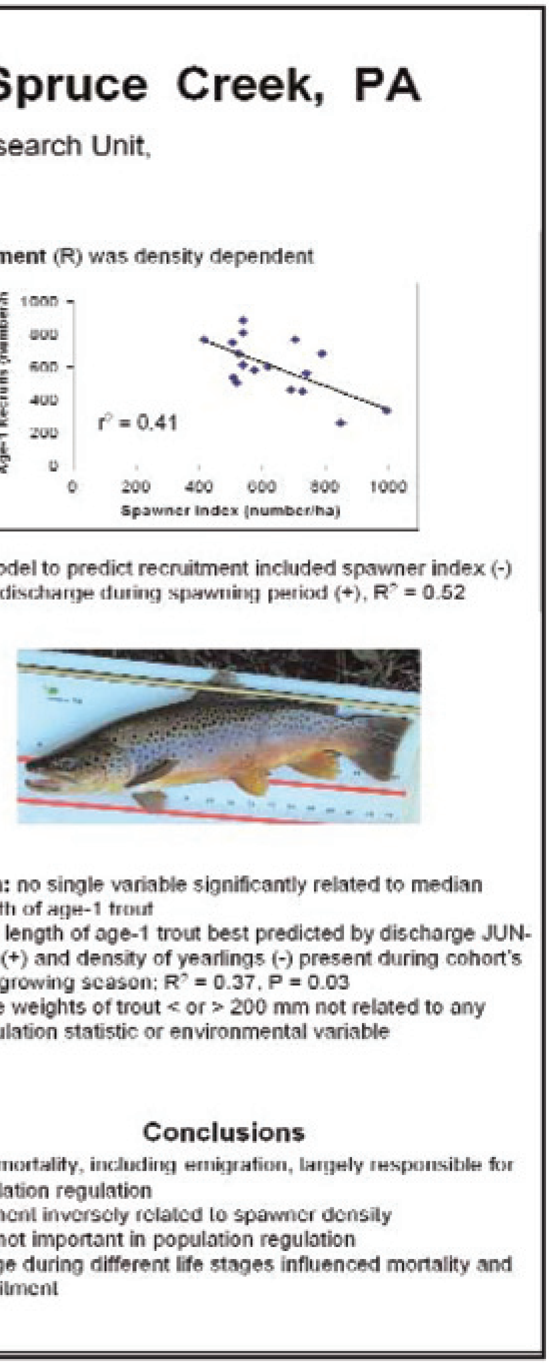
Posters have become an important mode of presentation at AFS Annual Meetings in recent years, because some topics are best communicated via posters and the number of requests to make oral presentations has far exceeded the available space and time. Thus, the number of posters has increased each year, and this trend will likely continue. Hundreds of posters may be on display, but the time available for meeting attendees to view them may be limited. When asked about their preferences for poster formats, meeting attendees were strongly in favor of posters that had a minimum of text and could be read in a relatively short time. These observations have prompted AFS to develop these new guidelines, which represent a significant deviation from previously recommended formats.

The purpose of the poster should be to convey highlights of a study or project in an attractive format that can be easily read and comprehended in a short period of time, i.e., 3 to 5 minutes. The body of the poster will have 300 to 400 words. In contrast, delivery of a 15-minute oral presentation may include 1,500 to 2,000 words, and it would include many more images than could be displayed on a poster. Thus, it is likely that a poster will convey less total information than that in an



oral presentation. Efficient use of this limited number of words and images is necessary to convey the highlights of the study.

A key feature of the poster is that it can be easily read at a distance of 2 m. Authors will need to minimize the amount of text in the poster, and to do so, use of bulleted phrases rather than complete sentences is best. Graphs need to be carefully designed so that they are readily comprehended. Details should be kept to a minimum. Photographs and color should be used to enhance the attractiveness of the poster and to entice the audience to stop and read it.



ELEMENTS OF THE POSTER

Title:

☐ The title should be short and fit across top of poster on one line.
☐ Authors' names and affiliations appear below the title.

Abstract:

☐ This section is optional.
*See below for additional information.

Introduction:

☐ Keep this section short.
☐ Limit it to a few statements.
☐ Clearly state the objectives.

Methods or Experimental Design:

☐ Keep text to a minimum.
☐ Use graphics where possible.

Results:

☐ This section should take up most of the space.
☐ Graphs (figures) are preferred over tables.
☐ Keep graphs simple.
☐ Include captions with graphics.
☐ Include credits on photographs taken by someone other than the authors.
☐ Tables should not exceed four columns.
☐ Keep statements brief.

Conclusions or Implications:

☐ Limit this section to a few bulleted statements.

References:

☐ This section is rarely included.

Acknowledgments:

☐ Include this section when appropriate.

***Abstract Option:**

A 200-word abstract in 28-point font will require 10% of the available space. Authors may decide that this space could be more effectively used for other material. Rather than require authors to include an abstract on the poster, this section is optional. If authors decide not to include an abstract on the poster, they should be sure to clearly state key items such as objectives and conclusions. Authors must provide the abstract as a handout at the poster location if the abstract is not included on the poster. Authors might also consider including a black-and-white (or color) reproduction of the poster on this same handout.

DESIGN SPECIFICATIONS

Overall size:

☐ The typical size of a poster is 91 cm x 112 cm (36" x 44") in a landscape or portrait format, but be sure to adhere to instructions from organizers of the specific meeting because their display boards may be better suited for posters of a different size.

Column arrangement:

☐ A 3-column format best fits this size poster in landscape format.
☐ The flow of material should be from top to bottom of each column and left to right among columns.
☐ Deviations from this pattern require careful planning.
☐ Leave 3.8 cm (1.5") between columns.

Highlighting the sections:

☐ One can use thin-lined borders around sections or blocks of subsections to emphasize how items are grouped.
☐ Light-colored background fill can also be used to highlight different sections.

Photograph backgrounds:

☐ Use of photographs as backgrounds is not recommended, because legibility is usually compromised.
☐ Text boxes with a background fill can be superimposed on photographs.
☐ Text printed directly on photographs should be avoided.

Background:

☐ Light pastel backgrounds are attractive and allow use of contrasting type colors, such as black, dark blue, and red.
☐ White backgrounds are acceptable, though they are less attractive than colored ones.

Font type and size

☐ Sans serif typefaces such as Arial are best for good visibility at a distance; use the same font type throughout.
☐ Title—72 point or larger; keep it short, not more than 80 characters including spaces.
☐ Authors' names and affiliations—48 point.
☐ Section headings—36 point, bold.
☐ Text—28 point.
☐ Graphs and tables—all numbers and labels 28 point or larger.
☐ Graph bars and symbols—use colors; avoid cross hatching.
☐ Acknowledgments—20 to 24 point.

Portraits of Authors

☐ Authors are encouraged to insert their pictures in the upper right corner of the title line. These portraits will assist poster viewers to find you, should they want to discuss your work.

JOB CENTER

To see more job listings go to www.fisheries.org and click Job Postings.

Chairperson and Professor, Department of Natural Resources Management, Texas Tech University/ Lubbock.
Responsibilities: The department includes the diverse disciplines of conservation, fisheries, range, and wildlife management. The chair must be familiar with these disciplines and have an understanding of the unique opportunities to integrate them into a multidisciplinary program. The department chairman has leadership and administrative responsibilities for the cepartment, and has the opportunity to teach and conduct research. Administrative duties include providing leadership and coordination of departmental teaching, research, and outreach activities; personnel and budgetary management; and liaison within the college and with agency and industry groups.
Qualifications: Qualifications include demonstrated leadership and administrative capabilities, experience in higher education teaching, and an established record of research productivity. A Ph.D. in conservation, fisheries, range, wildlife, or other closely related field is required.
Closing: 1 August 2007.
Contact: Applicants should apply online (Requisition # _73935_) at http://jobs.texastech.edu.

Biometrician I/II, Alaska Department of Fish and Game/Juneau (Douglas office) .
Responsibilities: Ensuring the biometric quality and planning for about 10–15% of the research projects in the Division of Sport Fish.
Qualifications: Biometrician I: MS in biometrics, statistics, or biological sciences including 18 semester/27 quarter hours of graduate level statistics or biometrics. Biometrician II: Same education plus 1-year experience at the Biometrician I level.
Salary: $3,664-$4,221 per month
Closing date: August 27, 2007
Contact: Allen Bingham, allen_bingham@fishgame.state.ak.us.
This position will be hired at either the Biometrician I or II level. To be considered at both levels separate applications are required. Application must be made with an Applicant Profile at http://workplace.alaska.gov/, and with Job Qualification Summaries for the two separate Recruitment Bulletins for Biometrician I or Biometrician II, Position ID Number: 11-5046.

Faculty Research Assistant, Oceanographic Data Analyst, Oregon State University, Cooperative Institute for Marine Resources Studies, Hatfield Marine Science Center/ Newport.
Responsibilities: This research project is a collaborative study of the relationships between environmental, climate, and oceanographic variability affecting annual fluctuations in the recruitment of marine fishes.
Qualifications: B.S. with experience, M.S. preferred in environmental, computer science, or fisheries oceanography science; strong quantitative skills with experience using statistical software packages to perform current and retrospective data analyses.
Salary: $36,000–39,600.
Closing date: 23 July 2007.
For full details see http://jobs.oregonstate.edu, Posting # 0000956.
Contact: Michael Schirripa, michael.schirripa@noaa.gov, or Jessica Waddell, jessica.waddell@oregonstate.edu for questions.

M.Sc. and Ph.D. Assistantships in Evolutionary Ecology, Department of Biology, University of Western Ontario.
Responsibilities: Research the evolutionary ecology of reproductive timing and senescence in Pacific salmon, involving field work on sockeye salmon and/or kokanee in British Columbia, laboratory work, and evolutionary modeling. Admission date of January 2008 or later.
Qualifications: Enthusiastic and goal-oriented individuals who pay close attention to detail, have relevant field experience (ability to work in inclement weather and rugged, bear-infested terrain, fish capture and handling

EMPLOYERS: To list a job opening on the AFS Online Job Center submit a position description, job title, agency/company, city, state, responsibilities, qualifications, salary, closing date, and contact information (maximum 150 words) to jobs@fisheries.org. Online job announcements will be billed at $350 for 150 word increments. Please send billing information. Listings are free for Associate, Official, and Sustaining organizations, and for Individual members hiring personal assistants. If space is available, jobs may also be printed in *Fisheries* magazine, free of additional charge.

techniques, boating skills), analytical skills (mathematics, statistics, computer programming), and a background in evolutionary theory. A minimum of 80% in your last two years of study.
Salary: Minimum $18,000 per year comprised of teaching assistantship and summer stipend.
Closing date: August 2007.
Contact: Send a one-page cover letter indicating your qualifications, current C.V., contact information for three referees, and unofficial copies of academic transcripts to Yolanda Morbey, ymorbey@uwo.ca. Applications will be screened on a continuous basis.

Senior Fisheries Biologist, HDR Inc., Anchorage, AK.
Responsibilities: Plan, direct and oversee all aspects of large scale, multi-discipline fisheries projects; provide oversight of field study program design and implementation for a wide variety of projects including fisheries assessments, fish population analyses, baseline studies, habitat improvement, and restoration; oversee advanced fisheries data analysis and provide quality assurance/quality control; build and maintain client relations; participate in project development and contract document preparation; and mentor mid- and junior-level fisheries biologists. This position will require field work in remote areas of Alaska for 1–2 weeks at a time.
Qualifications: B.S. in fisheries or related field, M.S. preferred. Fifteen plus years experience. Experience designing and directing large, complex, multi-discipline fisheries projects, including management of field studies.
Contact: Apply online at www.gojobs.com/seeker/aoframeset.asp?JobNum=1044026&JBID=1334. Employer JobCode: 061860.

2007 Membership Application

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Address ______________________
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City ____________ State/province ____________
Zip/postal code ____________ Country ____________

Please provide (for AFS use only)
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Fax ____________
E-mail ____________
Recruited by an AFS member? yes__ no__
Name ____________

Employer
Industry ____________
Academia ____________
Federal gov't. ____________
State/provincial gov't.__
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MEMBERSHIP TYPE (includes print *Fisheries* and online Membership Directory)	**North America/Dues**	**Other Dues**
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Developing countries II	N/A	$25 ______
Regular	$76 ______	$88 ______
Student (includes online journals)	$19 ______	$22 ______
Young professional ________ (year graduated)	$38 ______	$44 ______
Retired (regular members upon retirement at age 65 or older)	$38 ______	$44 ______
Life (*Fisheries* and 1 journal)	$1,737 ______	$1,737 ______
Life (*Fisheries* only, 2 installments, payable over 2 years)	$1,200 ______	$1,200 ______
Life (*Fisheries* only, 2 installments, payable over 1 year)	$1,000 ______	$1,000 ______

JOURNAL SUBSCRIPTIONS (optional)

Journal name	**North America** Print	Online	**Other** Print	Online
Transactions of the American Fisheries Society	$43 ____	$25 ____	$48 ____	$25 ____
North American Journal of Fisheries Management	$43 ____	$25 ____	$48 ____	$25 ____
North American Journal of Aquaculture	$38 ____	$25 ____	$41	$25 ____
Journal of Aquatic Animal Health	$38 ____	$25 ____	$41 ____	$25 ____
Fisheries InfoBase		$25 ____		$25 ____

PAYMENT Please make checks payable to American Fisheries Society in U.S. currency drawn on a U.S. bank or pay by VISA or MasterCard.
Check ____________ P.O. number ____________
Visa ____________ MasterCard ____________ Account #____________ Exp. date ________ Signature ____________

All memberships are for a calendar year. New member applications received January 1 through August 31 are processed for full membership that calendar year (back issues are sent). Those received September 1 or later are processed for full membership beginning January 1 of the followig year.
Fisheries, June 2007

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