Fish Bypass Flows for Coastal Watersheds

A Review of Proposed Approaches for the State Water Resources Control Board

Peter B. Moyle
Department of Wildlife, Fish and Conservation Biology
University of California Davis

G. Mathais Kondolf
Department of Landscape Architecture and Environmental Planning
and Department of Geography
University of California Berkeley

With Technical Assistance From
John G. Williams
Bay-Delta Modeling Forum

June 12, 2000

1. Introduction

In the Russian River watershed increasing pressure to develop hillside agriculture (especially vineyards) has led to a proliferation of water rights applications for diversions from headwater streams, which support federally listed coho salmon or steelhead, or support larger streams that do. Similar conditions occur in other coastal watersheds. The State Water Resources Control Board (SWRCB) is presently wrestling with the issue of how to condition permits for water rights to protect ecological resources, a task made difficult by the lack of information on the physical and ecological functioning of these channels, and their influence on downstream channels. For example, proposed methods for determining minimum instream flows in these streams have been developed using stream gauge data - all of which are from larger channels downstream, where scale differences lead to a very different hydrology. Similarly, the need for streamside protection zones along these headwater channels is not widely recognized, because most guidance has been developed for larger channels. In any case, existing institutions are poorly suited to regulating activities that impact these streams. The State Board can decide how much water (if any) should be diverted but has limited authority to regulate land use changes that influence runoff and erosion rates. Similarly, the Department of Fish and Game can put conditions on activities within the stream itself, but has limited authority beyond the stream banks. Land-use decisions are made at the county level, with varying levels of scientific analysis and political concerns influencing decisions. The most advanced county-level ordinance in the region is the Napa County Conservation ordinance, which is now under review in part because of concerns over its effectiveness in addressing the effects of multiple headwater impacts. Moreover, there is presently no mechanism for taking cumulative effects into account. The SWRCB has proposed analyzing cumulative hydrologic changes from numerous headwater diversions at the upstream point used by anadromous fishes, but this limit is changing in many streams as human-made barriers (such as culverts) are corrected as part of watershed restoration programs.

This review is intended to provide the SWRCB with guidance regarding minor water rights applications on streams in coastal watersheds, with particular focus on the Russian River basin. Many of these streams support coho salmon or steelhead rainbow trout, which are listed as threatened under the federal Endangered Species Act (ESA). Although there is general agreement that there is little if any water available for diversion in the dry season, frequent winter flooding in the Russian River basin supports the view that water could be diverted in some winter months without harmfully affecting instream flows required by salmon, steelhead, and other public trust resources. The SWRCB staff has developed an approach that, when embodied into permit conditions, is designed to allow for a "negative declaration" under the California Environmental Quality Act (CEQA) for diversions from small coastal steams; that is, a finding that exercise of a new permit will not have a significant effect on the environment. In other words, the conditions of each permit are supposed to be strict enough so that the diversion will not have negative effect on salmon, steelhead, or other significant aquatic life, either individually or cumulatively. Such findings have been made for several water rights applications in the basin of the Navarro River (SWRCB 1998), which supports coho and steelhead, and the SWRCB staff proposes to use the same approach in the basin of the Russian River, which also supports both species (SWRCB 1997).

The approach has been controversial, however, and has been criticized by the National Marine Fisheries Service (NMFS), Trout Unlimited (TU), the California Sports Fishing Protection Alliance (CALSPA), and others. This caused the SWRCB staff to seek review of the approach by qualified experts acceptable to the various parties, and the authors of this review were selected. The SWRCB also secured the services of the Executive Director of the Bay-Delta Modeling Forum, of which the SWRCB is an institutional member, to act as staff for the review. As part of the review, the SWRCB staff conducted a workshop, on 31 January 2000, in which the SWRCB staff approach and alternatives suggested by NMFS and TU were presented. The California Department of Fish and Game, CALSPA, and engineers from two private firms who frequently represent applicants for water rights, Wagner & Bonsignore and Napa Valley Vineyard Engineering, also participated in the workshop and provided comments.

In this review, we do not recommend a definitive method for determining what flows should be left in each stream to the SWRCB staff and interested parties. Instead, we give our views on topics raised in the workshop and related issues, give suggestions for improved formulation of permit conditions instream flow standards that are well suited for adaptive management, and recommend an approach to apply within the context of adaptive management.

2. General comments on instream flow standards:

Scientific uncertainty and Adaptive Management:

The implications of uncertainty for public policy and environmental management have been a topic of discussion in the scientific literature for some time (e.g. Holling 1978), but particularly in the last decade (e.g., Ludwig et al. 1993; Hilborn and Peterman 1995; Mangel et al. 1996; Chrisentsen et al. 1996; Francis and Shotton 1997; Healey 1997). The discussions have concerned environmental management generally, and management of fisheries or fish populations in particular, and have been motivated by rampant management failure: many stocks of commercially important fish have recently collapsed (Thompson 1993; Horwood 1993), and many runs of Pacific salmon and steelhead are either extinct or in trouble (Nehlsen et al. 1991; Stanley et al. 1996; Mills et al. 1997; Brown et al. 1994; Yoshiyama et al. 2000).

Briefly stated, it is now generally recognized among professionals that management of wild living resources involves such large amounts of uncertainty that (1) management actions are experiments and should be treated as such, and (2) irreversible actions should be avoided. This point of view is embodied in the widely advocated approach of "adaptive management" (e.g., Holling 1978; Walters 1986; Volkman and McConnaha 1993; Healey and Hennessey 1994; Healey 1997, Williams 1998). In 1996, we joined others in declaring that "currently no scientifically defensible method exists for defining the instream flows needed to protect particular species of fish or ecosystems," and in calling for the application of adaptive management to the problem of setting instream flow standards (Castleberry et al. 1996), with a focus on flows below existing dams. We made three basic recommendations:

First, conservative (i.e., protective) interim standards should be set based on whatever information is available, but with explicit recognition of its deficiencies. The standards should prescribe a reasonable annual hydrograph as well as minimum flows. Such standards should try to satisfy the objective of conserving fishery resources, the first principle of adaptive management (Lee and Lawrence 1986).

Second, a monitoring program should be established and should be of adequate quality to permit the interim standards to serve as experiments. Active manipulation of flows, including temporary imposition of flows expected to be harmful, may be necessary for the same purpose. This element embodies the adaptive management principles that management programs should be experiments and that information should both motivate and result from management actions. Often, it also will be necessary to fund ancillary scientific work to allow more robust interpretation of monitoring results.

Third, an effective procedure must be established whereby the interim standards can be revised in light of new information. Interim commitments of water that are in practice irrevocable must be avoided.

Here, we expand on these ideas, particularly as they relate to diversions from small streams.

The fact of relevant scientific uncertainty is perhaps best illustrated by recent developments in stream ecology. The role of high flows in structuring food webs in streams like those under consideration here has been elucidated only in the past decade (Wootton et al. 1996). Understanding of the substantial ecological importance of subsurface (hyporheic) flow, which is affected by the frequency with which stream sediments are mobilized, has also developed rapidly over the same period (Jones and Mulholland 2000). Although a great deal is known about salmonids and about stream ecosystems, these examples show that we should expect more surprises, and not assume that our current understanding is sufficient to support permanent decisions regarding management of streams.[1]

In adaptive management, uncertainty is acknowledged, management actions are recognized as experiments, and developing new information is an explicit management objective that can justify actions that may be sub-optimal in terms of other objectives. Deliberate manipulation of the system is required when there is otherwise little variation in the factor of management concern. For example, an adaptive approach to evaluating flows in a regulated stream with fairly constant flows would require a deliberate change in management; it is impossible to learn much about the relation between flow and public trust benefits in a stream if the flow does not vary. In other situations, an adaptive approach may not require deliberate manipulation of the system in question. Delta outflow in the spring, for example, varies naturally much more from year to year than it could from any plausible deliberate manipulation of outflows. The key in such situations is to describe the conceptual model upon which management is based as explicitly and quantitatively as possible so that the rationale for the standards can be formulated as testable hypotheses, and to establish a monitoring program by which the hypotheses can be tested.

---

[1] The great complexity of ecosystems and the practical impossibility of accurately measuring many relevant aspects of them explains the apparent paradox that scientists know a great deal about ecosystems but remain unable to make good specific predictions about how they will behave in response to small or intermediate perturbations. See Healey (1997) for a good discussion of this point.

Thus, adaptive management of instream flows may or may not require deliberate, experimental manipulation of flows, depending on the amount of variation that occurs in flows regardless of management. Generally, there will be large variation in flows within and between years in Russian River tributaries and in other California coastal streams. However, to depend on natural variation in flow for management "experiments" increases the risk that results will be confounded by other variables. For example, water quality might be better in high-flow years, so that benefits of improved water quality could be mistakenly interpreted as results of some hypothesized flow-habitat relationship. In any event, the rationale for the instream flow standards or permit conditions must be made clear, so that it can be tested against new information.

This can best be done if objectives and conditions or standards are stated in terms of explicitly biological criteria, with a method specified to convert these into hydrological terms. This allows the condition or standard to be articulated as a testable hypothesis or set of hypotheses. In the present context, for example, a winter flow standard or by-pass condition might be stated as a flow that allows enough spawning to occur to saturate the rearing capacity of the stream, stated quantitatively as enough flow to allow spawning to occur in 75% of the potential spawning habitat in the stream. To make this criterion operational it could be translated, based on some explicitly stated reasoning or evidence, as some particular value on the flow duration curve or some other parameter of the flow data. The standard or condition now involves two hypotheses, one harder to test than the other, but both at least conceptually testable. The more easily testable hypothesis would be that the selected flow criterion actually does allow for spawning in 75% of the potentially available habitat. The more difficult hypothesis would be that 75% of the potential spawning habitat will provide the desired level of biological protection, say lack of harm to listed species. The conceptual model in this case would be density-dependent effects such that spawning on 75% of the potential spawning habitat would saturate the rearing capacity of the stream, so that making more spawning habitat available would not result in greater production of juveniles or returning adults. In any event, the reasoning behind the standard or condition should be spelled out, so that is possible to specify the kind of information that would justify a change.

Under adaptive management, in other words, management decisions should *invite* change, by emphasizing uncertainty, by making clear what kind of new information would justify a change in the management action, and by requiring monitoring that can provide the relevant information. We emphasize this to clarify the difference between adaptive and traditional management, in which management actions typically are designed to be durable, and the reasoning given for the decision may be deliberately vague to further that end. Formulations such as "Careful consideration of all the evidence leads to the conclusion that a by-pass standard of X cfs best balances the competing needs for water," without further elaboration, are incompatible with adaptive management.

Similarly, the large scientific uncertainty regarding instream flows means standards or conditions must be based on explicit conceptual models and formulated as testable hypotheses. To depend on consensus of technical experts for the parties or stakeholders in a given situation, without these elements, may be convenient for decision-makers but damaging to the resource. Consensus on conceptual models and testable hypotheses would be very useful, but if the technical experts cannot articulate their recommendations in this way, the consensus is most likely based on non-scientific considerations, and as noted by Mangel et al.(1993) this approach has often failed:

> We believe that a principal reason for the routine overexploitation of resources is that the scientific community often fails to differentiate between science and policy, that is, to separate fact and value judgements. For example, scientists are often expected to reach "consensus" amid considerable scientific uncertainty about cause and effect. Instead of telling policy makers that they cannot accurately predict the consequences of alternative management strategies, scientists allow themselves to be forced into negotiated agreement. As a result, decision makers (usually not scientists themselves) are often not fully aware of the uncertainties and cannot be help fully accountable for the consequences of their actions.
>
> The International Whaling Commission, for example, asks its Scientific Committee to recommend catch quotas. Available information is often insufficient to determine catch levels that can be sustained, and many Scientific Committee members have different views about what should be done in the face of uncertainty; some believe that, when there is uncertainty, the benefit of the doubt should be afforded to the industry while others believe that it should be afforded to the resource. Instead of reporting the uncertainty and the possible consequences of this uncertainty to the Commission, the Committee generally has sought a "scientific consensus" that represents a middle ground. In hindsight, the consequence of attempting to reach a consensus is clear: one stock after another of the world's large whales have been driven to economic and near biological extinction.

In the current context, uncertainty in estimates of flows in small, ungaged streams is a major problem. The SWRCB has tried to address this problem through development of a rainfall-runoff model for the Russian River watershed, but the accuracy of this or any other model is fundamentally constrained by the scarcity of data on rainfall and runoff, which can be highly variable spatially in coastal watersheds. We make recommendations for addressing this problem below.

Other Types of Uncertainty and Other Factors to Consider:

Experience with fisheries management has demonstrated that uncertainty regarding non-scientific factors also needs to be taken into account for effective management, and doubtless the same is true for management of diversions from streams. Most obviously, uncertainty regarding compliance with permit conditions must be taken into account,[2] and the SWRCB should avoid allocating water to uses that would suffer seriously in dry years when permit conditions would limit diversions, unless it can assure compliance with the conditions. Uncertainty regarding future diversions under riparian rights, or expansions of diversions under appropriative rights, should be taken into account in such situations. Stated differently, effective management needs to take human motivation into account (Ludwig et al. 1993). In many situations, including the approach

[2] We appreciate the frank comments by Wagner and Bonsignore and Napa Valley Vineyard Engineering on this point.

under review, uncertainty about existing diversions under riparian or pre-1914 rights will be important, as will illegal diversions. Similarly, the SWRCB needs to consider the indirect effects of water allocations on streams. For example, if a small diversion from a headwater stream makes possible a use that will be accessed through the winter by a dirt road, then sediment from the road may have a greater effect on the stream than the diversion itself. Simply depending on other agencies to control such effects puts the puts the public trust at undue risk. Effective management needs to deal with the world as it is, not as it is supposed to be, and not as it is bounded by agency jurisdictions.

Limitations on Adaptive Management for Minor Water Rights Applications:

Water rights granted for vineyard development or other capital-intensive activities are for practical purposes irrevocable and their environmental effects should be evaluated in that light. This reduces the applicability of adaptive management to the process under review. Nevertheless, adaptive management still has a role, because much of the concern about the minor water rights applications involves cumulative impacts, so that future modification of the process for evaluating individual permits, in light of new information, can still be effective. However, the practical irrevocability of such allocations of water creates a greater need for caution than would otherwise be the case.

The practicality of effective monitoring of the efficacy of conditions on minor water rights permits also limits the applicability of adaptive management in such cases. Effective monitoring is almost always expensive, and the cost per unit of water diverted will be particularly high for small diversions. It seems to us that this difficulty can best be overcome by monitoring the effectiveness of permit conditions on a sample of diversions, with some method for spreading the cost over all diversions. Requiring inadequate monitoring of all diversions would be a waste of resources.

The need to protect high flows and flow variability:

The importance of maintaining high flows and flow variability seemed to be recognized by all parties at the workshop. We agree. There has been a spate of recent articles that emphasize the importance of variation in flow in rivers for creating and maintaining aquatic and riparian habitat and ecosystems (e.g., Ligon et al. 1995, Power 1995, Reeves et al. 1995, Sparks 1995, Power et al. 1996, Stanford et al. 1996, Wootton et al. 1996, Richter et al. 1997, Nilsson et al. 1997). As stated in the abstract of Power et al. (1996):

> Responses of rivers and river ecosystems to dams are complex and varied, as they depend on local sediment supplies, geomorphic constraints, climate, dam structure and operation, and key attributes of the biota. Therefore, "one-size-fits-all" prescriptions cannot substitute for local knowledge in developing prescriptions for dam structure and operation to protect local biodiversity. *One general principle is self-evident: that biodiversity is best protected in rivers where physical regimes are the most natural. A sufficiently natural regime of flow variation is particularly crucial for river biota and food webs.* We review our research and that of others to illustrate the ecological importance of alternating periods of low and high flow, of periodic bed scour, and of floodplain inundation and dewatering. The fluctuations regulate both the life cycles of river biota and species interactions in the food webs that sustain them. Even if the focus

> of biodiversity conservation efforts is on a target species rather than whole ecosystems, a food web perspective is necessary, because populations of any species depend critically on how their resources, prey, and potential predators also respond to environmental change. ... (Emphasis added.)

Brian Richter and his colleagues at the Nature Conservancy have developed an approach to evaluating instream flows from this point of view (Richter et al. 1996, 1997, 1998), although they acknowledge that the approach only provides a "first cut" that should be implemented in the context of adaptive management. The approach involves comparing up to 33 statistics developed from observed or simulated daily flow records for "project" and "no project" conditions, to develop and "index of hydrologic alteration," or IHA. A computer program to perform the analysis is available. The approach is strictly empirical, however.

Issues of Spatial Scale:

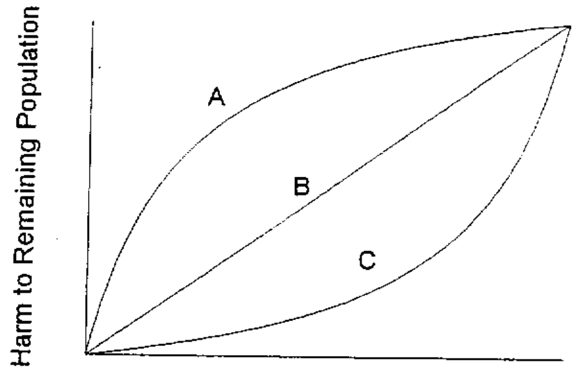
Issues of spatial scale are important in several aspects of the problem under consideration, as emphasized by TU. As one example, flows that provide adequate depth for migration of adult salmonids or for spawning become less frequent as the drainage area decreases. As another example, the flow in a stream reflects the integrated effects of rainfall over the basin, which may be highly variable if area of the basin covers more than a few square miles, so flow in the lower reaches is less variable than flow in the smaller tributaries. Therefore, applying hydrological generalizations developed from gage data to headwater streams is perilous, since gages typically are located in the lower reaches of stream systems.

At another level, there is ordinarily a need to balance instream and consumptive uses of water.[3] This balancing needs to be conducted at an appropriate spatial scale, however, if it is to be effective. Any such balancing in Russian River tributaries, for example, must place in the balance the amount of habitat that is blocked by Warm Springs Dam or otherwise degraded. This will create equity concerns on the part of water rights applicants on less modified streams, but meeting these concerns at the expense of the remaining habitat is a recipe for environmental disaster.

The equity concern just described raises an important question: how should we regard the incremental effect of additional habitat degradation in an already degraded system? The figure below shows three conceptual alternatives: Curve A reflects the idea that if an environment is already highly degraded, then a little more damage won't hurt much. Curve B shows a linear relationship, in which the harm to remaining members of a population does not depend on the general level of degradation, while Curve C reflects the idea that a high level of degradation makes any remaining habitat even more important. Of course, these curves grossly simplify a highly complex situation, but in our experience people do tend to evaluate evidence in terms of such simple conceptual models, so it useful to make them explicit.

---

[3] Our impression is that where federally listed species are involved, the balancing has already been done by Cong ress, but this point may still be relevant for tributaries too small to support salmon and steelhead.



For the situation at hand, we think that Curve C is most appropriate, although it should be regarded as a rebuttable presumption. In other words, the burden should be on applicants to show that Curve C is not appropriate. One reason for this is the essentially irrevocable nature of appropriative water rights, which makes the effects of choosing the wrong curve asymmetrical. For example, if Curve A really is the correct conceptual model, then acting as if Curve C is correct and denying a permit will result in temporary economic loss, since the water could be allocated after Curve A is shown to be appropriate. On the other hand, if Curve C is correct, then the consequences of issuing permits on the assumption that Curve A is correct will be serious and cause permanent harm to the population. This kind of asymmetry of effects, together with the scientific and other types of uncertainty described above, is the basis of the "precautionary principle" for fisheries management (Cameron and Abouchar 1991; Hilborn and Peterman 1996; Gordon and Munro 1996; Richards and Maguire 1998).

Legal barriers to rational water management:

California water law is a curious patchwork that has evolved in response to changing conditions in the state. Although it is possible to understand how the law came to be as it is, the law is nevertheless ill-suited for coping with the difficult allocation problems now facing the state. To scientists such as ourselves, for example, the different legal treatment of surface water and groundwater is fundamentally irrational and seriously compromises the state's ability to deal with its water problems. This is true at the state-wide scale at which we are currently advising CALFED, and it is also true at the scale of minor tributaries of the Russian River. The Public

Trust Doctrine provides the SWRCB with a powerful tool for accommodating appropriative water rights with protection of the public trust. However, demand in an area may be supplied partly by surface water diverted under riparian rights, partly by surface water diverted under appropriative rights, partly by "small domestic" certificates,[4] and partly by pumping of groundwater that is non-jurisdictional but is hydrologically linked to the surface streams. In such cases, developing rational and equitable conditions to impose on the exercise of appropriative rights is a task that we do not envy. The SWRCB is, in effect, working with one foot and one hand tied behind its back. Especially given the presence of listed species in a basin, the inability of the state to control effectively all water use within a basin means that even greater caution should be exercised regarding water use that the SWRCB can control than might otherwise be the case.

Existing unauthorized diversions:

The presence of many unauthorized diversions, some of long standing, creates a dilemma for the SWRCB. On the one hand, effective government depends upon the consent of the governed, and taking too strong a position against people who honestly do not realize that they need a permit for their diversions is likely to be counterproductive. On the other hand, taking too weak a position invites non-compliance, and deals with the problem at the expense of the public trust. We are not confident that there is a good resolution to this dilemma, but a vigorous program to identify unauthorized diversions and bring them into the water rights process would be an important step in the right direction. If the problem is ignored it will only get worse.

3. General comments on approaches discussed:

The SWRCB staff is attempting to develop an approach that, when embodied into permit conditions, will allow for a finding that the project in question will not have a significant effect on the environment. Under the ESA, harm to listed species is by definition a significant effect, so for the Russian River basin the approach must also allow for a finding that the project will not harm coho salmon or steelhead. Given the depressed condition of the populations of salmon and steelhead in the basin and the our limited knowledge of these fish and the ecosystems that support them, a finding of "no harm" can only mean that there is an acceptably low risk of a significant effect on the environment or harm to listed species. Reasonable people can differ in their assessment of what is an acceptable risk of harm. We emphasize, however, that given the condition of the stocks, the "reasonable range" of assessments includes the view that the Russian River and its tributaries are already over-appropriated, that existing diversions should be cut back, and that no new diversions should be allowed.

Although the condition of coho and steelhead populations the Russian River basin and elsewhere in California justifies particular attention to the effects of water diversions on these species, it bears emphasizing that the need for protection does not end with anadromous fish. In regions with Mediterranean climates, much of the drainage network is composed of intermittent headwater streams that flow seasonally. These channels support a distinct biota that has received

[4] Individual small domestic diversions can adversely affect very small tributaries even when they are not abused, and we suspect that abuse is not uncommon. Small domestic diversions also raise serious concerns regarding cumulative impacts. We think § 1228 et seq. of the Water Code should be reconsidered.

little attention, but deserves protection in its own right.[5] The seasonal streams may be particularly important for the breeding by amphibians (especially frogs), which are declining worldwide.

Such channels also convey water, sediment, organisms, organic litter, and large woody debris to perennial reaches downstream. As noted in Welsh et al. (2000), at the conclusion of a discussion of the critical role of large woody debris for pool formation:

> ... In a natural stream with intact riparian forests, a large proportion of these logs would enter streams from the highest channels in the stream network ... during large storm events (Sedell et al. 1998). Because they provide large woody debris and a variety of sediment types, headwater or first-order stream channels strongly influence the type and quality of downstream fish habitat (Sedell et al. 1998). Stated succinctly, "Reaches that are themselves inhospitable to salmonids may contribute to the maintenance of salmonid populations downstream" (G. Reeves in Reid 1998).

The ecological linkages between small headwater streams and the larger streams farther down the watershed mirrors the cumulative impact problem with minor diversions. Just as small headwater streams combine to form the larger streams that support anadromous fish, so many small diversions, which individually may be inconsequential, can combine to contribute substantially to the degradation of the stream system as a whole.

In some cases seasonal streams are even used directly by anadromous fish. For example, Trush (1991) observed that steelhead trout may ascend seasonal streams during winter freshets, spawn, and descend before flows drop below the minimum level for adult passage Their eggs hatch and the alevins emerge and migrate downstream in the spring before the channel dries up. Some juvenile chinook salmon in the Sacramento drainage also use seasonal tributaries for rearing habitat, and the same may be true of steelhead and coho salmon in the coastal streams.

Although most consideration has been given to steelhead and coho adults and to flows needed for spawning, winter habitat for juveniles is a major factor limiting recruitment for coho in coastal streams in Oregon (Nickelson et al. 1992) and British Columbia (McMahon and Hartman 1989; Hartman et al. 1996). The importance of winter habitat for juveniles in California is poorly understood but it clearly deserves more attention than it has received.

Finally, It seems to us that applying a single, "one size fits all" approach to instream flow standards for Russian River tributaries and other headwater streams in coastal watersheds is ill advised. The more general the approach, the more margin for error is required to support a finding of no significant effect. At the least, a distinction should be made between diversions from perennial streams or seasonal streams that carry continuous flow for part of the rainy season in most years, on one hand, and ephemeral streams or swales that flow only during or shortly after storms on the other. We discuss these separately below.

---

[5] See Gasith and Resh (1999) and Welsh et al. (2000) for recent reviews.

4. Proposed conditions on diversions from perennial or seasonal streams:

Summary of proposals:

The SWRCB staff proposed standard conditions that include three restrictions on diversions: (1) the season of diversion is restricted to December 15 to March 31; (2) the maximum rate of diversion is restricted, as determined on a case-by-case basis; and (3) diversions must allow a by-pass flow of 60% of the estimated mean annual unimpaired flow at the site. SWRCB permit terms in the Navarro watershed also include the provision that water diverted under claim of riparian rights not be used in the same area as water diverted under the permit, and we understand that this fourth constraint would apply elsewhere as well. NMFS agrees with the general form of the SWRCB staff proposal and the proposed season of diversion, but maintains that the by-pass standard should be the February median daily unimpaired flow, and that total diversions from a stream be limited to 20% of the 20% exceedence flow. TU also finds the basic form of the staff proposal and the season of diversion acceptable, but proposed that the by-pass flow be the 10% exceedence flow (90th percentile) daily unimpaired flow, that by-pass flows allow a minimum passage depth of 0.8 to 1.0 ft, and that total diversions from a stream not advance the recession of storm hydrographs to the by-pass flow by more than 0.5 to 2 days, depending on the size of the watershed. Wagner and Bonsignore and Napa Valley Vineyard Engineering also found the basic form of the staff recommendation acceptable, although impractical and overly burdensome in some specifics. At the workshop, there seemed to be convergence of opinion toward the general limit to total diversions proposed by NMFS.[6] In summary, there is agreement regarding the basic approach, but differences regarding several of the specifics of its implementation.

We are not persuaded that it is wise to issue any new permits until effective recovery programs for coho salmon and steelhead are in place, but with that caveat we also find the general form of the approach acceptable, and agree that a hydrologically-based approach is reasonable provided that the hydrological criteria are explicitly linked to biological criteria by testable hypotheses. The form of the NMFS proposal for limiting total diversions seems reasonable, although we have not evaluated the specific criterion that NMFS has proposed. Effective implementation of this approach would require knowledge of all existing legal and illegal diversions, however, for which data are largely lacking at present. We also agree with NMFS that negative declarations are inappropriate for proposals for impoundments on perennial or seasonal streams. Such impoundments are likely to have a significant effect on the environment, even if conditions requiring by-pass flows are made part of the permit. Apart from concerns regarding compliance with by-pass requirements, such impoundments will drown stream habitat that has ecological value even if it does not support fish, and will effect other stream habitat by interfering with the migration of organisms and downstream movement of sediment and organic matter as well as water. We also agree with TU that a separate minimum depth criterion may be necessary, particularly for smaller streams.

SWRCB staff proposal for by-pass standards:

The 60% mean annual flow by-pass flow proposed by the SWRCB staff is based largely several PHABSIM studies that indicate that 60% of the mean annual flow will provide 80% of "weighted usable area" (WUA) for coho and steelhead spawning. During the workshop, the SWRCB staff clarified that their proposed by-pass flow is intended to allow for substantial

[6] This approach is detailed in pp. 28-29 in NMFS (2000).

spawning, and not just to provide holding habitat between high flow events during which spawning might take place.

In form, this recommendation is close to what we think is needed. That is, there is a biological objective, and the approach is based on a conceptual model (which underlies PHABSIM) that can easily be formulated as testable hypotheses. We cannot endorse this standard, however, for several reasons, some of them raised in comments by NMFS and TU. A first reason concerns scale effects: some minimum depth is required for adult passage and spawning, but the depth provided by some fixed percentage of the mean annual flow will decrease with the watershed area. Accordingly, applying the results of studies on relatively larger streams to smaller ones is dubious. A second reason concerns the uncertainty associated with the results of any method for estimating spawning habitat, and the presumed dome-shaped relation between flow and spawning habitat in a given stream. Even if PHABSIM results involved relatively little uncertainty, small underestimates of the flow that would produce 80% of maximum spawning habitat could produce relatively large reductions in the actual spawning habitat, particularly because the flow-habitat curves tend to be steeper to the left of the selected point.[7] In other words, the SWRCB staff approach does not provide an appropriate margin for error.

More seriously, the uncertainty in the results of PHABSIM studies is very large. PHABSIM is based on the premises that habitat value of a point in a stream can be described in terms of the depth, water velocity, and the substrate, and that the area of a reach of stream with given values of depth, velocity and substrate can be estimated using hydraulic models. The descriptions are based on "preference" or "suitability" curves that vary between 0 and 1 as a function of depth, velocity, and substrate, using different curves for different life stages. The hydraulic modeling is normally done with one-dimensional models, which describe the stream in terms of a set of transects, as was the case with the studies cited. Problems with PHABSIM using one-dimensional hydraulic modeling are described by Williams (1996) and Kondolf et al. (in press, attached as an appendix), and references cited therein. Briefly, there is a good deal of uncertainty in model results at the transects, and much more uncertainty from extending results at the transects to the rest of the stream. In terms of spawning, there is a clear additional problem with the conceptual model underlying PHABSIM: salmonids select spawning sites partially in terms of "hyporheic" or subsurface flow, so depth, velocity and substrate do not adequately describe spawning habitat.

In short, we believe that the PHABSIM studies cited by the SWRCB staff report do not provide an appropriate basis for by-pass conditions or flow standards, so the 60% criterion is essentially arbitrary. This does not mean that the 60% criterion is necessarily wrong, but rather that it lacks a suitable proximate rationale against which it could be judged. It seemed to us, however, that the discussions in the workshop and in the NMFS comments (p. 16) raised serious questions about the adequacy of the 60% criterion to avoid harm to spawning by steelhead and coho salmon, especially in smaller tributaries.

Finally, as noted above, we are concerned about the uncertainty in estimates of mean annual flow (or estimates of any point on the flow-duration curve) from the streamflow simulation model, that is proposed for use as part of the SWRCB staff approach. Probably a good deal of

[7] See Figure 4.1-2 in Attachment B to SWRCB (1997) for an example.

this uncertainty is unavoidable; precipitation in mountainous areas is highly variable temporally and spatially, and gages tend to be concentrated in more populated areas at lower elevation and relief. Measurements of stream flow from gage data are more accurate than estimates of areally-averaged precipitation, but 95% confidence intervals for flow measurements at gages are probably about +/- 5%, so even with measured data there is some uncertainty. Although we are not rainfall-runoff experts and have not carefully reviewed the model, it also seems to us that uncertainty in the estimates will increase as the size of the basin under consideration decreases, so the tests of the model presented in the SWRCB's 1997 Russian River Watershed Staff Report (errors of 7.6 and 10.3%) most likely underestimate the errors that should be expected when the model is applied to smaller areas.[8]

NMFS Proposal:

The February-median flow by-pass standard proposed by NMFS is based on two considerations: that more flow (within some limit) provides more spawning habitat, on the one hand, and that the flow must be sustained for a considerable period for the spawning to be successful, on the other. NMFS finds that the February-median flow is an easily defined criterion that reasonably balances these considerations, or in other words that the median February flow approximates the flow that will maximize the habitat in which coho salmon and steelhead can successfully spawn. NMFS also assumes that maximizing the effective spawning habitat will maximize production of steelhead and coho salmon (i.e., survival of juveniles is not strongly density-dependent, at least given current population levels). These assumptions can easily be cast as hypotheses, so the NMFS proposal is consistent with the form that we recommend. The first hypothesis, that the February median flow approximates the flow that maximizes effective spawning habitat, would be much easier to test than the second.

The NMFS criterion is more conservative than the 60% of mean annual flow standard proposed by SWRCB staff, and as noted above a more conservative approach is appropriate. Given the status of salmon and steelhead in the Russian River basin, and the absence of a realistic recovery plan, it is reasonable to maintain maximum spawning habitat in tributaries that do or could support these fishes, until good evidence is developed to show that less spawning habitat is required. This is particularly appropriate for an approach that is intended to allow for use of negative declarations under CEQA,.

NMFS also proposes that the cumulative diversion at any point on a stream not exceed 20% of the "winter 20% exceedence flow," following a procedure outlined at p. 28 in their comments, for which "winter" means December 15 to March 15. As noted above, there seemed to be convergence towards this approach in discussion at the workshop, and with the caveat noted

[8] We are also concerned by the statement at the end of Section 5 in Attachment A of the Staff Report that the model results were more variable when it was used with rainfall and runoff data for the same period (e.g., 1961-1981 for Macama Creek). We are not sure we that understand this statement, but it raises questions in our minds about the model testing. We also do not understand why the model tends to shift peaks in the average weekly flow data forward in time, especially later in the year (Figures 5 and 6 in Attachment A to SWRCB 1997) but this raises more questions. It seems to us that the model is really more of an empirical model than a physically-based model, and that explicitly empirical regression models might do as well or better for the intended use.

above it seems reasonable to us, although we have not done independent analyses of the specific criterion. Presumably NMFS agrees with the SWRCB staff that the maximum rate for individual diversions should be determined on a case by case basis.

Trout Unlimited Proposal:

The proposal by Trout Unlimited (TU) is also described in terms of hydrology, although two of the three criteria proposed are explicitly linked to biology. As described in the 1/10/00 letter from Bill Trush to Jerry Johns, TU proposes that:

(1) "No streamflow between December 15 and March 31 should be diverted below a stage height equivalent to the 10% daily average flow exceedence (p) on an unimpaired daily average flow duration curve."

(2) By-pass flows should allow a minimum passage depth of 0.8 to 1.0 ft (which will be more restrictive than (1) in smaller watersheds).

(3) In any stream, diversions should not advance the recession of storm hydrographs to the base flow determined (1) or (2) by more than 0.5 to 2 days, depending on the size of the watershed.

According to Trush's letter of 1/10/00, criterion 1 is "...associated with an hydraulic break in the channel's hydraulic geometry and is readily identified in the field as a morphologically distinct inner channel." This is also described as the "active channel" in the McBain and Trush commentary of 3/12/98, identifiable by "(1) the lower limit of rooted mature white alders, (2) the crest of an abrupt berm along the outer margin of bars, and (3) a bench of finer alluvium along glide and riffle margins." The commentary also summarizes observations regarding use of the active channel by steelhead from Trush's graduate research (Trush 1991). Criterion (2), regarding depth of flow, would be converted to a specific discharge by means of a relationship between depth and drainage area that Trush is developing under a contract with NMFS.

A basic difficulty with the TU proposal is that criteria (1) and (2) are based upon observations that have not been described in the peer- reviewed literature, and have not been subjected to ordinary professional scrutiny. We have reviewed materials provided to us by Trush (Trush 1991, Trush undated) and find that they would not persuade a skeptical reader that there is a morphologically distinct inner channel that is corresponds to the area occupied by the 10% exceedence flow in his study area, Elder Creek. Such an inner-channel may well exist, but the evidence for it has not yet been presented effectively. In any event, the generality of Trush's Elder Creek observations would need to be established before they would provide a reasonable basis for regulation.[9]

---

[9] The active channel shelf feature identified reported by McBain and Trush (1998) and Trush (1991) has been reported from other river systems in the peer-reviewed literature. As noted in Trush (1991), Osterkamp and Hedman (1977:256) described the active channel shelf as:

> ...a short-term geomorphic feature subject to change by prevailing discharges. The upper limit is defined by a break in the relatively steep bank slope of the active channel to a more gently sloping

TU's third criterion raises an important point that should be considered before a specific total limit on diversions in the form proposed by NMFS is adopted; diversions will reduce the duration of flows greater than the by-pass standard, as well as the magnitude of such flows. With a small enough storm, a diversion could remove a flow pulse entirely, so the criterion as proposed may not be workable, but we think this point should be evaluated in some quantitative way, for example by use of the IHA software (Richter et al. 1996; 1997), as well as by visual evaluation of "with project" and "without project" hydrographs.[10]

5. Proposed Impoundments on ephemeral streams or swales:

For the reasons described above, the SWRCB staff should use caution and judgement in approving impoundments on ephemeral streams,[11] but in many situations this may be acceptable,

---

surface beyond the channel edge. The break in slope normally coincides with the lower limit of permanent vegetation so that the two features, individually or in combination, define the active channel reference level.

While the features appear to be the same, the frequencies of inundation are reported to be rather different. From a study of channel geometry at 70 gauging stations on mostly intermittent or ephemeral streams in the semi-arid western US, Hedman and Osterkamp (1982:3-4) reported these relations between the active channel and flow regime:

> At most perennial and intermittent streams the active channel level is exposed between 75 and 94 percent of the time. The active-channel level of many ephemeral streams may be exposed more than 99 percent of the time. The stage corresponding to mean discharge of most perennial streams approximates that of the active-channel level ... but is lower than the active channel level of the highly ephemeral stream channels...

In the (perennial) Passage Creek drainage basin in Virginia, Hupp and Osterkamp (1986) found that the active channel shelf was inundated between 5-25% of the time, and supported a riparian-shrub forest.

Thus, while the association of the active channel feature with the 10% exceedence level in north coastal California channels proposed by Trush is plausible, results in the published literature suggest considerable variation in the percentage of time that the active channel shelf is inundated. Scale issues are important. As shown by Hupp (1986), as one goes headward along a drainage, features like the floodplain and then the active channel shelf disappear completely. Thus, the relevance of the 'active channel' in headwater streams needs to be confirmed before being adopted as a basis for establishing instream flows there. At the least, the applicability of the return periods and exceedence levels observed on larger channels to headwater channels is questionable. As Trush (1991) pointed out, "The case study of Elder Creek main channel morphology and steelhead spawning ecology has a sample size of one. Conclusions derived from monitoring and hypothesis testing cannot be statistically extrapolated to other drainage's or to tributaries within the Elder Creek Watershed" (p. 72).

Kondolf and Williams have observed the active channel shelf feature on many coastal California streams, but in some cases it was clearly the result of deposition of debris flow material brought in by steep tributaries. It is not clear to us why such deposits should be related to any particular point on a flow-duration curve, rather than the particular conditions existing just after the debris flow.

[10] People tend to underestimate differences represented by a pair of sloping lines because the normal distance between the lines is much easier to see than the more significant vertical and horizontal distances.

[11] We recognize that the SWRCB does not have jurisdiction over impoundments that capture "diffuse surface waters."

and in some cases it may be necessary to allow storage from diversions from larger seasonal or perennial streams. We agree with the suggestion made by CDFG during the workshop that there must be a limit to the percentage of a watershed controlled by impoundments, although there remains the question from which point to calculate this percentage. Clearly, 100% of the watershed above each impoundment will be so controlled, and the percentage will decrease moving downstream from each dam, unless there is a confluence with a more heavily regulated stream. Probably there is no rigid formula that will make sense in all cases. One possible approach would be to specify the limit in terms of a percentage of the watershed of first order streams, with recognition that there will be areas, for example swales that drain directly into second or higher order streams, to which this formula would not sensibly apply. The effects of these impoundments on high flows downstream should also be taken into account in estimates of total diversions and limits on cumulative diversions.

We recommend that impoundments only be permitted under negative declarations only when "fill and spill" operation is acceptable, so that permit compliance issues are minimized.[12] More flexibility regarding the season of diversion may also be appropriate for such cases, so that the effects of different diversions can be distributed temporally.

Additionally, we recommend a requirement that impoundments be emptied annually, for two reasons. The first and most important reason is that perennial ponds provide habitat for exotic species such as bullfrogs. The danger from these exotics far outweighs any incidental or opportunistic use of such ponds by native species, including listed natives. Secondly, a requirement that ponds be emptied will greatly facilitate compliance monitoring; a pond will either be effectively empty before the allowed season of diversion, or it will not.

6. Minimum level of analysis:

Even with conservative bypass standards, field investigations will always be necessary to provide the information necessary for a Negative Declaration. More importantly, the SWRCB can learn whether its permit conditions adequately protect public trust resources only if it has information regarding current conditions to which future conditions can be compared. We recommend that one set of field investigations be used for both purposes. We have reviewed the negative declarations prepared for several Navarro River and Russian River applications, and find that the level of analysis is less than is needed. Although any rigid formula for field investigations is likely to be burdensome for some cases and inadequate for others, we think a typical field investigation probably should include the following:

Reconnaissance survey: After inspecting topographic maps and recent aerial photography,[13] SWRCB staff or DFG staff should walk the channel from the project site downstream to the confluence with a substantially larger stream (unless the diversion is directly from a stream known to be easily accessible to salmonids) to detect and evaluate unusual conditions that call for special

---

[12] For example, we are concerned about compliance problems with by-pass conditions such as those proposed for Application No. 29711, because it appears that inflow to the impoundment will be much less than capacity in dry years, when the need for the water will be greatest.

[13] Aerial photography is readily available from commercial sources, and applicants should be required to submit images of the project area and the affected reach of stream as part of the application.

treatment. For example, a waterfall that partially blocks fish migration may make upstream diversions even of high flows problematic, since the high flows may be needed to allow passage over the barrier. We realize that securing access may be a problem, but this burden can be placed on the applicant. We do not see how a finding of no significant impact can be made if the affected reach of stream cannot be inspected.

Photodocumentation: Channel conditions should be recorded by photographs showing both typical and unusual conditions. The photographs should be annotated using notes made during the reconnaissance or other field visits.

Discharge measurements: SWRCB or CDFG staff should measure the discharge in the stream whenever they visit a project site. Even one or a few discharge measurements can provide an important check on calculated estimates of flow. If the discharge is less than about 3/4th to 1 cfs the measurement should be make using a portable flume; if it is larger, current meters should be used. Measurements made between storms during the season of diversion will be most valuable, and if possible field visits should be scheduled to allow for them.

Channel characteristics: SWRCB staff should characterize the channel geometry near the project site and downstream. This should include sketched channel transects, with dimensions estimated using a staff or tape, measurements of slope[14], and estimates of channel roughness. These should be used to estimate stage over a range of discharges, to provide a check on the plausibility of calculated estimates of flow at the site, and to provide a baseline description of the channel to allow for future assessments.[15] If there are sites such as bridges that provide convenient sites for future measurements that can show incision or aggradation, then more care should be taken in depicting the transect accurately at these sites. Channel substrate should be described, using quantitative methods such as pebble counts (Kondolf 1997) where they are appropriate..

Vegetation: Vegetation in the project area, especially riparian vegetation, should be characterized and common species should be listed. Exposed roots or drowned trees that reflect channel incision or aggradation should be recorded, as should stands of even-age riparian trees, the elevation of flood scars on riparian trees, or other features that provide evidence regarding stream processes.

Characterization of aquatic fauna: Perennial stream should be examined at least twice, once in late summer at minimum flow and once in winter when spawning salmonids are likely to be present. Seasonal streams should be examined in late winter or early spring. The wetness or dryness of the year should be taken into account.[16] Direct sampling of fish (e.g., electrofishing) should be used if possible; at the least observations should be made of the presence or absence of

[14] Adequate measurements of slope can be with a hand level in steeper streams (say >2% slope), but an auto level should be used for streams with lower slopes; the slope should be measured over a distance of at least 10 channel widths.

[15] Problems with simple before/after comparisons, described in Schmitt and Osenberg (1996), need to be kept firmly in mind, but probably there is no practical way to avoid them in the present context.

[16] Ideally, streams should be inspected twice in both wet and dry years. As an alternative, appropriately sized streams in the same area could be inspected in a space for time substitution.

fish (species if possible), presence of redds, or other evidence of fish using the steam. Presence of amphibians (adults and larvae) should also be noted. Invertebrate communities should be characterized using CDFG's rapid bioassessment procedure or some other procedure that identifies the abundances of major aquatic taxa. It is important that careful, standardized notes be taken at each note, preferably on a special form.

The success of field investigations depends critically upon the skill, experience and attitude of the investigator. No methodology, procedures, checklists or forms to fill out can substitute for the ability to "read" streams and associated landforms. Similar skills are required to assess whether the proposed diversion as constrained by the by-pass conditions makes economic sense, or whether there will be an unacceptably large motivation to cheat. Essentially this means that to be successful, the SWRCB must be able to maintain competent staff and provide for their continuing education.

In the negative declaration, the analysis of the amount of water available at the site should be reported in enough detail (probably in an appendix) to allow others to repeat the calculations, and should describe the assumptions of the method used and how well the assumptions are met at the site in question. Put differently, in order that the assumptions of the method be testable, the method used needs to be described well enough that it can be checked against discharge measurements in the stream, should such measurements be made in the future. In any event, the main body of the study should include an assessment of the likely accuracy of the reported estimates, and field conditions should be used to check the plausibility of the estimates.[17] The analysis should also include a discussion of the availability of water during severe drought as well as of a typical dry year, since the project is most likely to have a significant effect on the environment during severe droughts, and uncertainty regarding compliance with permit conditions will also be greatest.

7. Comments on monitoring and research:

Estimates of the flows that should be expected in ungaged tributaries is a major source of uncertainty that could be reduced substantially by a well-designed monitoring and research program. Developing the design for such a program is beyond the scope of this review, and should involve knowledgeable people for agencies such as the USGS, NRCS, DWR, and county or local agencies, as well as academics. The SWRCB should take the initiative in promoting the design and implementation of such a program, and it should be willing to exercise its power to re-

---

[17] We are concerned about the methods used in the Navarro River basin negative declarations to estimate the amount of water available at the project sites. Without data, no method will be very accurate, so it is appropriate to use a simple method. Making reasonable estimates with such methods requires considerable skill knowledge and experience with the region in question, to guide selection of parameters for the model; simply plugging in numbers for a table can lead to gross errors. It is also important that the method not be biased. The initial studies refer to the Rational Method, which is intended to predict peak flows. It is not clear to us what method was used for estimating average annual flows. Unfortunately, such methods for predicting peak flows are intended for sizing culverts or similar applications where the harm from underestimates is much greater than the harm from overestimates, so the methods are biased high. For estimating the amount of water available for appropriation, or the amount that will be left in the stream, a bias in the opposite sense is appropriate.

open existing permits to add conditions needed for implementing the program. Future permits should include requirements for collecting and reporting precipitation and flow data, although the specific requirements should be tailored to individual cases.[18]

Since making assessments of the availability of water for proposed projects is a routine part of the SWRCB's work, however, the SWRCB should have strong in-house expertise in this area. Based on the SWRCB documents that we have reviewed, this expertise is currently lacking. Therefore, we recommend that the SWRCB create a staff position at a sufficiently high level to attract an individual with demonstrated knowledge and experience in this area. This person would also represent the SWRCB in the development of the coordinated monitoring and research program described above, and participate actively in its implementation.

As with hydrological uncertainty, research and monitoring intended to address the biological uncertainties involved in assessments of the effects of water diversions should be coordinated with other efforts, if this is possible. A better understanding of the biology of coho salmon, steelhead, and the coastal streams that support them is also needed to address important issues regarding timber harvest, for example, and this understanding could best be developed by a coordinated effort. Again, scientists from various agencies and from universities should to be involved, but the SWRCB can and should work for the creation of such a coordinated program.

Four biological topics stand out as requiring particular attention for testing the hypotheses implicit in the NMFS approach to conditioning permits and for reducing uncertainty about the environmental effects of diversions with such conditions: the use of streams by coho salmon and steelhead as spawning habitat; the nature of density-dependent mortality among juvenile salmon and steelhead; the use of streams as winter rearing habitat by these fishes, and characterization of ecosystems in seasonal or small perennial streams.

Trush's (1991) observations of steelhead spawning in Elder Creek, combined with geomorphically informed attention to channel conditions, exemplify the kind of work that is needed regarding spawning habitat. These need to be repeated in other streams, however, particularly because there is now greater awareness of the importance of hyporheic flow as an aspect of salmonid spawning habitat.

Observational studies are also needed of the use of winter rearing habitat by juvenile coho and steelhead. Studies of winter habitat use by salmonids in other areas should provide conceptual models and hypotheses to be tested in coastal California, but streams here are typically warmer in the winter and this should be taken into account. Winter habitat as been identified as a factor limiting survival of juvenile coho, so this topic overlaps with the general issue of density-dependent mortality among juvenile salmon and steelhead. This is a difficult issue but strong density-dependent mortality in the fry life stage has been demonstrated in anadromous brown trout (Elliott 1994), so the assumed lack of strong density-dependent mortality underlying the NMFS proposal needs to be examined carefully.

---

[18] A fee to help cover costs of the monitoring program could be substituted for data collection in some cases, especially in areas for which other data are available.

Studies of the ecosystems of seasonal and small perennial streams should be guided by the conceptual models and hypotheses that are already in the literature (e.g., Gasith and Resh (1999) and Welsh et al. (2000)), but there is also a basic need for simply characterizing the biota.

8. Summary and Recommendations

1. There is substantial uncertainty regarding the conditions needed to allow recovery of coho salmon and steelhead populations in coastal watersheds in California, and regarding the flow regime needed to maintain ecosystems in small headwater streams. There is also substantial uncertainty in estimates of the expected flow in streams at project sites, and about the actual effectiveness of mitigation measures prescribed by water right permits.

2. The historical decline and current status of coho salmon and steelhead populations, the pervasive modification of aquatic habitats in coastal watersheds in California, the unknown cumulative effects of legal and illegal diversions, and the scarcity of basic data on headwater streams are sufficient reasons to justify deferring approval of any new water rights, particularly in the Russian River watershed, until information is developed that shows that the diversions can be conditioned to avoid unacceptable risk of harm to listed species or other public trust resources.

3. If SWRCB feels obligated to approve diversions from seasonal or perennial streams using negative declarations, despite incomplete knowledge of both local and cumulative impacts of the diversions, we suggest using the NMFS approach, with the addition of a separate depth criterion for smaller streams that are used by anadromous fishes, and with consideration of the effects of diversions on the duration of high flows. In doing this, the SWRCB should confront uncertainty and pursue adaptive management by:

- Basing by-pass standards and flow requirements on clearly defined objectives;
- Using biological and hydrological criteria that can expressed as testable hypotheses;
- Requiring a monitoring program that can test the hypotheses; and
- Modifying standards in light of new information.

4. Impoundments should not be approved on seasonal or perennial streams using negative declarations. Impoundments should be approved on ephemeral streams using negative declarations only where a "fill and spill" approach is acceptable, and the impoundments should be emptied annually to control exotic species, especially bullfrogs.

5. The SWRCB should work with other state, federal and local agencies and academic institutions to promote improved hydrological and biological data collection and research to reduce the uncertainties identified above, and to test the hypotheses underlying management decisions and permit conditions. The SWRCB should develop a process whereby monitoring that is intensive enough to be effective can be focused on selected sites. The SWRCB should develop greater in-house expertise in estimating flow at the sites of proposed projects.

9. References:

Allan, J.D. 1995. Stream Ecology: structure and function of running waters. Chapman and Hall.

Benda, L.E. 1994. Stochastic geomorphology in a human mountain landscape. Doctoral dissertation. University of Washington, Seattle.

Brown, L. R., P. B. Moyle, R. M. Yoshiyama. 1994. Historical decline and current status of coho salmon in California. North American Journal of Fisheries Management. 14:237-261.

Cameron, J. and J. Abouchar. 1991. The precautionary principle: a fundamental principle of law and policy for the protection of the global environment. Boston College International & Comparative Law Review 14:1-27.

Castleberry, D.T., J.J. Cech, D.C. Erman, D. Hankin, M. Healey, G.M. Kondolf, M. Mangel, M. Mohr, P.B. Moyle, J. Nielsen, T.P. Speed, and J.G. Williams. 1996. Uncertainty and instream flow standards. Fisheries 21(8):20-21.

Christensen, N.L., A.M. Bartruska, J.H. Brown, S.R. Carpenter, C.D'Antonio, R. Francis, J.F. Franklin, AJ.A. MacMahon, R.F. Noss, D.J. Parsons, C.H. Peterson, M.G. Turner, and R.G. Woodmansee. 1996. The report of the Ecological Society of America report on the scientific basis for ecosystem management. Ecological Applications 6:665-691.

Connel, J.H. 1978. Diversity in tropical rain forests and coral reefs. Science 199:1302-1310.

Elliott, J.M. 1994. Quantitative Ecology and the Brown Trout. Oxford University Press.

Francis, R.I.C.C. and R. Shotton. 1997. "Risk" is fisheries management: a review. Canadian Journal of Fisheries and Aquatic Sciences 54:1699-1715.

Gasith, A. and V.H. Resh. 1999. Streams in Mediterranean Climate Regions: abiotic influences and biotic response to predictable seasonal events. Annual Review of Ecology and Systematics 30:51-81.

Geist, D.R. and D.D. Dauble. 1998. Redd site selection and spawning habitat use by fall chinook salmon: the importance of geomorphic features in large rivers. Environmental Management 22:655-669.

Gordon, D.V. and G.R. Munro, editors. 1996. Fisheries and uncertainty: precautionary approach to resource management. University of Calgary Press, Calgary, Canada.

Hartman, G.F., J.C. Scrivener, and M.J. Miles. 1996. Impacts of logging in Carnation Creek, a high energy coastal stream in British Columbia, and their implication for restoring fish habitat. Canadian Journal of Fisheries and Aquatic Sciences 53 (Supplement 1):237-251.

Healey, M. 1997. Paradigms, policies and prognostication about watershed ecosystems and their management. In Naiman, R.J. and R.E. Bilby (eds). River Ecology and Management: Lessons from the Pacific Coastal Ecoregion. Sppringer-Verlag, New York.

Healey, M.C. and T.M. Hennessey. 1994. The utilization of scientific information in the management of estuarine ecosystems. Ocean and Coastal Management 23:167-191.

Hedman, E.R., and W.R. Osterkamp. 1982. Streamflow characteristics related to channel geometry of streams in western United States. US Geological Survey Water-Supply Paper 2193. US Geological Survey, Washington DC.

Hilborn, R. and R. M. Peterman. 1996. The development of scientific advice with incomplete information in the context of the precacutionary approach. United Nations Food and Agriculture Organization Technical Paper No. 350/2:77-102.

Holling, C.S. (ed.) 1978. Adaptive Environmental Assessment and Management. John Wiley & Sons.

Horwood. J. Stochastically optimal management of fisheries. ICES CM D:26. ICES, Copenhagen.

Hupp, C.R. 1986. The headward extent of fluvial landforms and associated vegetation on Massanutten Mountain, Virginia. Earth Surface Processes and Landforms 11:545-555.

Hupp, C.R., and W.R. Osterkamp. 1985. Bottomland vegetation distribution along Passaage Creek, Virginia, in relation to fluvial landforms. Ecology 66:670-681.

Jones, J.B. and P.J. Mulholland. 2000. Streams and Ground Waters. Academic Press.

Kondolf, G.M. 1997.Application of the pebble count: notes on purpose, method, and variants. Journal of the American Water Resources Association 33(1): 79-87.

Lee, Kai N, and Jody Lawrence. 1986. Adaptive management: learning from the Columbia River Basin fish and wildlife program. Environmental Law 16:431-460.

Ligon, F.K., W.E. Dietrich, and W.J. Trush. 1995. Downstream ecological effects of dams. Bioscience 45:183-192.

Ludwig, D., R. Hilborn, and C. Walters. Uncertainty, resource exploitation, and conservation: lessons from history. 1993. Science 260:17,36; reprinted with responses in Ecological Applications 3:547-549.

Mangel, M., R.J. Hofman, E.A. Norse, and J.R. Twiss, Jr. 1993?, Sustainability and ecological research. Ecological Applications 3:573-575.

Mangel, M.C. and 41 others. 1996. Prinicples for the conservation of wild living resources. Ecological Applications 6:338-362.

McMahon, T.E. and G.F. Hartman. 1988. Influence of cover in habitat use by juvenile coho salmon (Oncorhynchus kisutch). Canadian Journal of Fisheries and Aquatic Sciences 46:1551-1557.

Mills, T.J., D. McEwan, and M.R. Jennings. 1997. Califoriia salmon and steelhead: beyond the crossroads. Pages 91-112 in D.J. Strouder, P.A. Bisson, and R.J. Naiman (eds), Pacific Salmon and their Ecosystem. Chapman & Hall.

NMFS National Marine Fisheries Service). 2000, Draft, Guidelines for maintaining instream flows to protect fishery resources in tributaries of the Russian River. Unpublished report, NMFS, Southwest Region, Santa Rosa, California.

Nehlsen, W., J.E. Williams, and J.A. Lichatowich. 1991. Pacific salmon at the crossroads: stocks at risk from California, Oregon, Idaho, and Washington. Fisheries 16(2):4-21

Nickelson, T.E., J.D. Rodgers, S.L. Johnson, and M.F. Solazzi. 1991. Seasonal changes in habitat use by juvenile coho salmon (Oncorhynchus kisutch) in Oregon coastal streams. Canadian Journal of Fisheries and Aquatic Science 49:783-789.

Nilsson, C., R. Jansson, and U. Zinko. 1997. Long-term responses of river-margin vegetation to water-level regulation. Science 276:798-800.

Osterkamp, W.R., and E.R. Hedman. 1977. Variation of width and discharge fornatural high-gradient stream channels. Water Resources Research 13:256-258.

Pickett. S.T.A. and P.S. White (eds.). 1985. The Ecology of Natural Disturbance and Patch Dynamics. Academic Press.

Power, M.E. Floods, food chains, and ecosystem processes in rivers. 1995. Pages 52-60 in C.G. Jones and J.H. Lawton. Linking species and ecosystems. Chapman and Hall, N.Y.

Power, M.E., W.E. Dietrich, and J.C. Finlay. 1996. Dams and downstream aquatic biodiversity: potential food web consequences of hydrologic and geomorphic change. Environmental Management 20:887-895.

Reid, L.M. and S. Hilton. 1998. Buffering the buffer. Pages 75-85 in R.Ziemer, ed., Proceedings of the conference on coastal watersheds: the Casper Creek story. USDA FS PS-GTR 165.

Reeves, G.H. L.E. Benda, K.M. Burnett, P.A. Bisson, and J.R. Sedell. 1995. A disturbance-based ecosystem approach to maintaining and restoring fresh-water habitats of evolutionarily significant units of anadromous salmonids in the Pacific Northwest. American Fisheries Society Symposium 17:334-349.

Richards, L.J. and J.J. Maguire. 1998. Recent international agreements and the precautionary approach: new directions for fisheries management science. Canadian Journal of Fisheries and Aquatic Sciences 55:1545-1552.

Richter, B.D., J.V. Baumgartner, J. Powell, and D.P. Braun. 1996. A method for assessing hydrologic alteration within ecosystems. Conservation Biology 10:1163-1174.

Richter, B.D., J.V. Baumgartner, R. Wigington, and D.P. Braun. 1997. How much water does a river need? Freshwater Biology 37:231-249.

Richter, B.D., J.V. Baumgartner, D.P. Braun, and J. Powell. 1998. A spatial assessment of hydrologic alteration within a river network. Regulated Rivers: Research and Management 14:329-340.

Schnidt, R.J. and C.W. Osenberg (eds.) 1996. Detecting Ecological Impacts: concepts and applications in coastal habitats. Academic Press.

Sedell, J.R., P.A. bission, F.J. Swanson, and S.V. Gregory. 1998. What do we know about large trees that fall into streams and rivers. Pages 47-81 in From the Forest to the Sea: a story of fallen trees, C. Maser, R.F. Tarrant, J.M. Trappe, and J.F. Franklin , editors. Pacific Northwest Research Station PNW-GTR-229, Portland, OR: USDA Forest Service.

Sparks, R.E. 1995. Need for ecosystem management of large rivers and their floodplains. Bioscience 45-:168-182.

Stanford, J.A., J.V. Ward, W.J. Liss, C.A. Frissell, R.N. Williams, J.A. Lichatowich and C.C. Coutant. 1996. A general protocol for the restoration of regulated rivers. Regulated Rivers, Research and Management 12:391-413.

Stanley, T.L., K.D. Hyattm, T.G. Northcote, and R.J. Fielden. 1996. Status of anadromous salmon and trout in British Columbia and Yukon. Fisheries 21(10):20-35

State Water Resources Control Board, Divsion of Water Rights. (SWRCB). 1998. Navarro River Watershed, Mendocino County, Draft Division Decision, Pending Applications 29711, 29810, 29907, 29910, and 19911.

State Water Resources Control Board, Divsion of Water Rights. (SWRCB). 1997

Thompson, G.G. 1993. A proposal for a threshold stock size and maximum fishing mortality rate. Pages 303-320 in Smith, S.J., J.J. Hunt, and D. Rivard, eds., Risk evalatuion and biological referenced points for fisheries management. Canadian Special Publications in Fisheries and Aquatic Sciences 120.

Trush, W.J. 1991. The influence of channel morphology and hydrology on spawning populations of steelhead trout in South Fork Eel River tributaries. PhD dissertation, Wildland Resrouce Science, University of California, Berkeley, California, 195 pp.

Volkman, J.M. and W.E. McConnaha. 1993. Through a glass darkly: Columbia River salmon, the Endangered Species Act, and adaptive management. Environmental Law 23:1249-1272.

Walters, C.J. 1986. Adaptive Management of Renewable Resources. McGraw-Hill, New York.

Welsh, H.H. Jr., T.D. Roelofs, and C.A. Fissell. 2000. Aquatic ecosystems of the redwood region. Pages 165-199 in The Redwood Forest, R.F. Noss (Editor). Island Press.

Williams, J.G. 1998. Thoughts on adaptive management. Interagency Ecological Program for the Sacramento-San Joaquin Estuary Newsletter 11(3):5-11.

Wootton, J.T., M.S. Parker, and M.E. Power. 1996. Effects of disturbance on river food webs, Science 273:1558-1561

Yoshiyama, R. M., E. R. Gerstung, F. W. Fisher, and P. B. Moyle. 2000. Chinook salmon in the California Central Valley: an assessment. Fisheries 25(2): 6-20.

Attachment 1

to

Fish Bypass Flows for Coastal Watersheds

A review of proposed approaches for the State Water Resources Control Board

Peter B. Moyle and G. Mathias Kondolf

Measuring and Modeling the Hydraulic Environment for Assessing Instream Flows

G. Mathias Kondolf
Dept. of Landscape Architecture and Environmental Planning
University of California, Berkeley CA 94720

Eric W. Larsen
Dept. of Geology, University of California
Davis CA 95616

John G. Williams[1]
875 Linden Lane, Davis CA 95616
jgwill@dcn.davis.ca.us

In Press
(2000 or 2001)
North American Journal of Fisheries Management

1. Author for correspondence

Key words: Instream flow, hydraulic models, velocity measurement, PHABSIM

Abstract

Detailed measurements of depth and velocity in natural channels, although rare, show that the velocity fields are complex and irregular even in streams with moderate gradients and gravel substrates. This complexity poses a challenge for instream flow studies, most of which use PHABSIM, a set of computer models that combine the results of hydraulic modeling, estimates of channel substrate or cover, and habitat suitability criteria to compute weighted usable area (WUA), an index of habitat. Some recent studies have replaced the transect-based one-dimensional (1-D) hydraulic modeling in PHABSIM with 2-D models that allow better definition of the depth and velocity fields in the modeled stream reach. The accuracy of the estimates as a function of channel geometry and data collection effort remains unclear, however, as does the utility of the estimates for evaluating instream flow needs. Here we review the assumptions, accuracy and precision of hydraulic modeling and of the measurements that provide input data for the models, and consider some implications of the consequent limitations of hydraulic modeling for describing fish habitat and assessing instream flows. Highly accurate hydraulic modeling seems unfeasible for streams with complex channel geometry, and in any event practical hydraulic modeling cannot resolve flow patterns at the short length scales at which fish often respond to the hydraulic environment. Information on depth, velocity, and substrate is important for assessing instream flows, but information developed from hydraulic models should be treated with great caution and is not a substitute for biological understanding.

Detailed measurements of depth and velocity in natural channels are rare, but those that do exist show that the velocity fields are complex and irregular, often with significant cross-stream components (Dietrich and Smith 1983, Petit 1987, Whiting and Dietrich 1991; Larsen 1995; Whiting 1997). This complexity in the flow patterns in natural channels poses a challenge for methods of assessing instream flows that depend upon hydraulic modeling, such as the Physical Habitat Simulation Model (PHABSIM).

PHABSIM consists of a set of computer models that combine hydraulic and biological models to evaluate the habitat value of a reach of stream for a given fish species and life stage. The weighted sum of calculated habitat values for the reach is expressed as "weighted usable area" (WUA), which is taken to represent the "living space" available for the organism; water quality and temperature are evaluated separately. PHABSIM is widely used in North America as a tool to quantify the biological effects of alternations in flow regimes or the relative habitat benefits of different flow release regimes from reservoirs (Reiser et al. 1989), and has increasingly been applied overseas as well, either directly or in modified form (Jowett 1989; Pouilly et al. 1995). PHABSIM has even been used to evaluate the instream flow needs of blue ducks (*Hymenolaimus malacorhynchos*), which forage for invertebrates in steep, boulder-bedded upland streams of New Zealand (Collier and Wakelin 1996). However, the hydraulic and biological aspects of PHABSIM have also been the subject of continuing criticism (e.g., Marthur et al. 1985; Shrivell 1986, 1994; Osborne et al. 1988; Gan and McMahon 1990; Elliott 1994; Castleberry et al. 1996; Ghanem et al. 1996; Heggenes 1996; Williams 1996; Lamouroux et al. 1998).

In this paper, we consider the adequacy of hydraulic models in general, and PHABSIM in particular, for making predictions of the depth and velocity fields in natural rivers that are useful for assessing instream flows. We begin with data from the literature that demonstrate the complexity of the depth and velocity fields in natural steams. We then consider the sampling and measurement problems associated with developing data for modeling the flow fields in natural channels, or for describing the flow fields empirically . We next consider modeling approaches, given practical restrictions on data collection. Finally, we consider some biological aspects of the problem, and offer some recommendations. We confine ourselves to the problem of estimating the habitat value of a stream for a single species and life stage of fish, although we recognize the inadequacy of that perspective for real environmental protection. We do not consider recently reported hydrologically-based methods for assessing instream flow regimes (Richter et al. 1996, Richter et al. 1997); these appear promising, but do not explicitly link physical characteristics of channels to flows or biological habitats.

Depth and Velocity Fields in Natural Streams

The data of Whiting and Dietrich (1991) illustrate the complexity of patterns in natural channels. They took detailed measurements on Solfatara Creek, a 5-m-wide gravel bed stream draining 62 km^2 in Yellowstone National Park, Wyoming. The 20 m-long study reach was located downstream of a bend, where the creek flows over and around a mid-channel bar; the substrate is coarse sand to medium gravel, and the average channel slope is 0.001 (Figure 1). Measurements were made at about one-third bankfull stage, using an array of small current meters suspended from a portable wooden bridge, across eleven cross sections spaced 2 m apart.

Although the stream appears relatively tranquil at this discharge, the velocity field is quite complex (Figure 2), displaying large variations vertically and horizontally within a given section, as well as between closely spaced sections. The large variation in channel form and velocity distributions from one section to the next, despite the close spacing of the sections, illustrates the spatial sampling problems inherent in any transect-based methods for evaluating instream flows. Results would vary substantially depending on the precise location of transects. Spatial sampling problems would be even more severe in steeper streams with larger substrate.

The measured velocity fields show that vertical velocity profiles often deviate substantially from the logarithmic profile commonly assumed (Figure 2), as has been noted elsewhere (e.g., Dingman 1989, Beebe 1996); in particular, the highest velocities are sometimes near the bed (e.g., cross sections 1 and 2). This implies that measurements of velocity at 0.6 depth may give only an approximation of the true column velocity. To illustrate this point, we obtained data for eight of the sections or transects shown in Figure 2 (not all data are available because of a storage media failure) and compared the vertically averaged velocity computed from measurements spaced 5 cm or less apart with the velocity at 0.6 depth (Figure 3). The velocity at 0.6 depth overestimates the vertically averaged velocity in most cases (the median difference is about +6%), but underestimates the vertically averaged velocity by almost 60% at some verticals in Section 10, where the flow deepens after passing over a mid-channel bar. In steep streams with large roughness elements, flow patterns would be even more complex. It may be possible to model the spatially averaged vertical velocity gradient in such streams (Wiberg and Smith 1991), but only if the stream is straight and the roughness elements are distributed approximately randomly, i.e., not organized into bars. These conditions are fairly restrictive, and as is often noted in discussions of instream flows, fish do not live in averages.

Details of the flow can vary in important ways even where general patterns are similar. This is illustrated in Sections 1-4, which have approximately the same shape and general lateral distributions of velocity, with higher velocity in the deeper part of the channel. Yet, the velocity gradients are quite different in Sections 1 and 2 compared to Sections 3 and 4. At Sections 1 and 2, the vertical gradient is almost nonexistent near the outside of the bend, but then becomes very steep under the high velocity core, which is near the bottom. Such steep gradients do not occur at Sections 3 and 4. If velocity gradients are important for fish, as indicated by the literature (e.g., Jenkins 1969; Bachman 1984; Heggenes 1994, 1996), then such differences would be important, but would remain undetected without detailed measurements of velocity and bed topography.

Note that the change in channel shape with distance downstream forces significant changes in the velocity field, termed convective accelerations. This has implications for modeling, because one-dimensional (1-D) models ignore convective accelerations.

Velocity Measurement in Streams:

For each cross section or transect measured at Solfatara Creek, Whiting and Dietrich took an average of 160 point velocity measurements, each a time-average over two minutes, requiring 8-10 hours to complete. In most practical applications, it is not possible to spend 8-10 hours per transect to measure velocity. PHABSIM procedures are typically modeled after the standard procedures of the US Geological Survey for measuring velocity in discharge measurements near stream gauges, described in Rantz et al. (1982).

Velocity is measured at 20-30 stations across the channel by wading or from a cable or bridge, using a Price AA current meter or the smaller mini current meter, consisting of cups that spin around a vertical axis in response to moving water. For depths less than 0.8 m, velocity is measured at 0.6 depth (i.e., 40% of the vertical distance from the bed to the water surface), which is assumed to reflect the mean column velocity. In deeper flow, the average of measurements at 0.8 depth and 0.2 depth is taken as reflecting the mean column velocity. The mean column velocities for each point are multiplied by the measured water depth and by the width of the vertical slice of the cross section represented by this measurement, to obtain the discharge for that vertical slice. The discharges for the individual "verticals" are summed to obtain the total discharge past the cross section.

To obtain a good measurement of flow, the hydrographer measures the stream by wading when possible, selecting the cross section with the most uniform flow conditions available on the channel, i.e., with flow lines that are parallel and that do not vary downstream. The hydrographer will often "improve the [measurement] cross section by removing rocks and debris within the section and in the reach of channel immediately upstream and downstream from the section," or by constructing "... temporary dikes to eliminate slack water...", all in an effort to transform flow conditions in the irregular natural channel into more uniform flow conditions (Rantz et al. 1982). Each measurement is rated as excellent, good, fair, or poor, with assumed error margins of 3%, 5%, 8%, or >8%, respectively, assigned based on the hydrographer's judgement (Rantz et al. 1982).

Ratings of "excellent" are uncommon in natural streams, despite the hydrographer's freedom to select the most uniform reach available and to modify channel geometry. The reaches selected for discharge measurements are probably not the preferred habitats for fish, or at least they are not typically the sites where anglers would look for fish. In essence, the hydrographer seeks the reach of channel that most closely resembles a canal. Highly irregular channels with shallow marginal areas, back eddies, still water, or boulder beds, which may be important as fish habitats, are sites that a hydrographer would avoid for flow measurement (unless the stream offered nothing better) because the resulting measurement would be poor.

Sources of Error in Measurements

Errors in point measurement of depth are usually small. At some locations the depth of flowing water can fluctuate by several centimeters at constant discharge, but this can be detected by reasonably careful observation of the section. Errors in estimating the average depth of a vertical are most likely to be sampling errors, especially when the cross section is irregularly shaped or the substrate is coarse. These conditions should be obvious, especially when measurements are made by wading, and with reasonable care a good estimate should be possible.

Potential sources of error in velocity measurements include the inherent limits of accuracy of the meter in registering downstream current velocity, temporal variations in velocity at a point, vertical and cross sectional components of velocity, and sampling errors within each vertical. Instrument errors associated with measuring unidirectional flow with Price meters are relatively minor; in the controlled environment of a tow tank, Carter and Anderson (1963) found that Price meters register within 0.6% of the actual downstream velocity. However, these meters were in excellent condition; poorly maintained meters, or meters clogged with sediment or organic debris, would not perform so well.

Replicate discharge measurements in rivers using Price and Ott current meters (a screw-type meter) were found to differ by up to 2.8% in total discharge (Carter and Anderson 1963). Agreement between the two meters seems acceptable, although the actual differences in point velocity measurements not reported. However, PHABSIM studies often use Marsh-McBirney current meters, which use the distribution of pressure around a rounded sensor to estimate velocity. This is conceptually attractive, and Marsh-McBirney meters can provide instantaneous or time-averaged readings of velocity. Manufacturer's specifications for the Marsh-McBirney meter state the accuracy as +/- 2% of reading, with a +/- 0.05 ft/s offset. Although one Marsh-McBirney meter performed well in initial tests by the US Geological Survey (Fulford et al. 1994), subsequent tests with a number of meters showed variable performance, under- and over-registering low velocities (Janice Fulford, US Geological Survey, pers. comm. 1998). In our experience the meters can be unstable and require frequent calibration, and after informal field comparisons with a Price current meter we are skeptical of data collected with Marsh-McBirney meters.

The vertical and cross-channel components of velocity are not well captured in the standard US Geological Survey flow measurement. The Price AA meter does not measure flow direction. Although any cross-channel flow can be accounted for using the hydrographer's estimate of the angle of approach, the existence of cross-channel flow at a vertical indicates a complex flow structure, so that one or two measurements may give a poor estimate of the spatially averaged velocity in the vertical. The Price meter is also affected by vertical velocity components in steep, turbulent channels but cannot measure them separately from the downstream components (Townsend and Blust 1960, Linsley et al. 1982). The velocities recorded in such channels may be greater than the true downstream velocities (Marchand et al. 1984). A modified Price meter that has solid cups composed of a polycarbonate polymer (The PAA meter) initially appeared to be less affected by vertical velocity components than the standard AA meter with stainless steel cups (Marchand et al. 1984), but subsequent experience has shown the polymer cups less accurate than the original stainless steel cups (R. Jarrett, U.S. Geological Survey, pers. comm. 1998).

There can be considerable temporal variation in velocity at a point in a stream, particularly one with a rough bed. The standard US Geological Survey approach is to take the velocity measurement over at least 40 seconds. Carter and Anderson (1963) took measurements continuously for an hour in 23 different rivers, at four different depths. They recorded data every 15 seconds, which allowed them to calculate the deviations of velocity measured over shorter intervals around the one-hour average (Figure 4). Although there are some problems with these data, they show that sampling errors are still significant at 40 seconds. Errors are also greater near the bed, where "focal point" velocity measurements are often made. Thus, the 40-second rule reflects a compromise between the gain in accuracy from averaging over a longer period and the cost of the additional time required. However, this compromise was developed for discharge measurements, where random errors in individual measurements tend to average out over the transect. In PHABSIM, measurements are not averaged over the transect, and it is not clear that the same compromise is appropriate. Moreover, the data are from reaches selected for discharge measurements, and greater temporal variation should be expected in reaches with more complex geometry.

Spatial sampling errors within each vertical will depend on the complexity of the flow field. In canal-like sections, the spatial sampling errors are small enough to allow good or excellent discharge measurements. In a complex flow field, however, even for a relatively tranquil stream such as that illustrated in Figures 1 and 2, the spatial sampling errors in estimating the average velocity of a vertical from one or two velocity measurements can be substantial.

Commonly, the discharge during a PHABSIM study is assumed to be known from a nearby gage, and if the total flow calculated by summing the individual PHABSIM measurements differs from the "known" discharge, the individual velocity measurements are adjusted by "velocity adjustment factors," which are percentage changes applied equally to all the measurements across the channel (Milhous et al. 1984). Although this adjustment may account for systematic errors, it does nothing to change the distribution of sampling and measurement errors across the channel.

In summary, instrument errors with well-maintained and properly used Price or Ott current meters are likely to be small, relative to temporal and spatial sampling errors. Figure 4 provides some guidance regarding temporal sampling errors. Although the figure probably underestimates the magnitude of the errors for transects with complex flow patterns, a similar decrease in the sampling error with increased measurement time can be expected. With standard methods, spatial sampling errors are probably as large or larger than temporal sampling errors. Herschy (1978) provides for a more detailed discussion of measurement errors at sites selected for discharge measurements, and gives "rules of thumb" for estimating 95% confidence intervals around measurements at such sites (Table 1). Unfortunately, there have been too few detailed studies of the flow field in natural channels to allow quantitative generalizations about measurement errors in channel reaches more typical of those to which PHABSIM is applied, rather than those selected by hydrographers for discharge measurements. For the conditions of most instream flow studies, however, we believe that the errors in estimating the average velocity of verticals by the standard methods will be large enough to affect ultimate results, so the ordinary scientific practice of estimating errors by appropriate repetitive measurements should be followed.

Modeling Flow in Natural Streams:

One-dimensional models:

One-dimensional (1-D) models typically treat the river as a series of cross sections, for each of which a stage and cross-sectionally averaged velocity are computed based on hydraulic principles, the channel form, and calculated values of stage and velocity at downstream cross sections. Probably the best-known 1-D model is HEC-2, or HEC-RAS, which is widely used for predicting flood levels. WSP, a similar 1-D gradually varied flow model, is an option for modeling stage in PHABSIM (Milhous et al. 1984).

One-dimensional models typically assume that the channel is straight, with all flow perpendicular to the cross section, and that flow is either "uniform" or "gradually varied". Uniform flow does not change in the downstream direction, and therefore has a vertical velocity profile that reflects a balance between the acceleration of gravity and the resistance of the channel bed. These conditions can occur in canals, but generally not in natural streams. "Gradually varied" flow occurs where channel topography and roughness change only slowly along the channel, so that convective accelerations can be ignored.

These are large assumptions, and while reasonable approximations of river stage are routinely obtained with these models if they are used with adequate skill and professional judgement, by definition they can provide only cross-sectionally averaged velocity. Moreover, gradually-varied flow models are commonly used for predicting flood stage during high flows. During such high flows, variations in the bed topography may be relatively less important; for example, hydrologists speak of riffles being "drowned out" at bankfull stage and higher. Whiting (1997) has shown that convective accelerations are less important at higher flows in Solfatara Creek. Instream flow assessments, however, are typically concerned with the lower magnitude flows in which fish spend most of their time. These flows are too low to modify the bed, so they occupy a channel geometry inherited from past high flows. Downstream changes in channel geometry that are small relative to high flows may be large relative to low flows, as when low flow spills over a longitudinal bar, so that the assumption of gradually varied flow is violated, as noted by Osborne et al. (1988). As a result, a model that gives reasonable estimates of stage in a channel at high flows may fail to do so at low flows.

PHABSIM is concerned with the distribution of velocity and depth across the channel, so the hydraulic models in PHABSIM divide the cross section into vertical slices (cells) either centered on or between point measurements of velocity (much as is done in the USGS discharge measurements). The vertical cells are analyzed separately, using either a regression analysis of measurements of velocity in the cell at different stages, or a back-calculation of Manning's n from a single velocity measurement (Milhous et al. 1989). The latter approach has been properly criticized by Shirvell (1986), and more recently by Ghanem et al. (1996), who point out that the cells are no longer tied to one another through hydrodynamic principles. On this account, Ghanem et al.(1998) describe the velocity modeling in PHABSIM as "zero-dimensional". With the single measurement approach, the Manning's roughness factor is used to calculate velocity and discharge for each cell at other discharges, but the individual cell discharges are adjusted to equal the modeled flow, so the roughness factor is really a weighting factor rather than a true roughness coefficient. With the multiple measurement approach, there is a problem with obtaining the required three velocities for verticals near the bank, which may be dry at the lower measured discharges (Ghanem et al. 1996).

Errors associated with the PHABSIM approach to distributing velocity across channels were investigated by Bartz (1990), as part of a broader assessment of PHABSIM, using data from the US Fish and Wildlife Service for three streams spanning a flow range of two orders of magnitude. For each stream, he calibrated different PHABSIM hydraulic models to data at three flows, and for each vertical compared the measured and modeled velocities. The averages and standard deviations of the differences are substantial, as illustrated by data for the medium-sized stream (Figure 5): mean errors ranged from 4.6% to 12.8% and standard deviations ranged from 29.6% to 42.7%. Results for the small and large stream are similar.

Two-dimensional Models:

Two-dimensional (2-D) models are increasingly being used for instream flow studies (e.g., LeClerc et al.1995, Ghanem et al. 1996). Two-dimensional models require the simultaneous solution of a system of governing equations, typically including relationships (expressed as differential equations) for conservation of fluid mass, conservation of downstream fluid momentum, and conservation of cross-stream fluid momentum. To simplify these relationships, certain approximations are assumed, yielding the "shallow water equations". These 2-D velocity models give only vertically-integrated velocities, but show the variation in cross-stream direction as well as in the downstream direction.

These models retain the convective acceleration terms neglected by 1-D models, but require more detailed descriptions of channel geometry, and the accuracy of the modeled results depends upon the accuracy and spatial resolution of the measurements (Leclerc et al. 1995, Ghanem et al. 1996). For example, Leclerc et al. (1995) constructed a computer representation of the bed of a large stream by measuring the bed elevation with one measurement for every 50 to 400 m^2, so their results are necessarily generalized accordingly.

However, with detailed specification of the channel bed topography and planform, more sophisticated modeling may not be necessary. One-dimensional models are not all the same, and in some settings 1-D models can be as accurate for simulating vertically integrated velocity fields as a 2-D approach. Dietrich (1987) modeled flow in Muddy Creek, Wyoming, for geomorphic purposes, with a 1-D approach that explicitly accounted for the effect of channel curvature, and predicted the distribution of velocity across the transects. Larsen (1995) applied the same 1-D approach, and compared observed velocity patterns on two gravel-and-cobble-bedded meandering rivers. He showed that, with good bed topography as input, the 1-D model performed as well as more sophisticated models. However, understanding the appropriateness and limitations of a model seems critical. For example, it is unlikely that the excellent results achieved by Dietrich (1987) and Larsen (1995) could be achieved in a straight channel with irregular bed topography, such as the reach of Solfatara Creek studied by Whiting and Dietrich (1991), for which a 2-D model that accounted for convective accelerations would be more appropriate.

Statistical Hydraulic Models

Following a suggestion by Dingman (1989), Lamouroux et al. (1995) developed an empirical model that predicts the statistical distribution of hydraulic variables (such as velocity and water depth) for reaches with intermediate and large roughness elements, for which they believe the conventional deterministic models are ineffective. The model predicts the distributions of the hydraulic variables over an entire reach based on inputs of discharge, mean width and depth, and roughness. Lamouroux et al. (1998) coupled this hydraulic model with multivariate habitat use models to estimate the habitat value of a reach as a function of discharge. The need for validation is perhaps more obvious with such straightforwardly empirical models, which is a virtue.

Model Validation:

Models by nature involve simplifications of reality, and model predictions always involve some error. For hydraulic modeling of fish habitat, the errors can arise from measurement errors, from model errors, or from sampling errors. With the standard 1-D versions of PHABSIM, one should ask how accurately depth and velocity were measured at the selected points on the transects, how well the model predicts depth and velocity at the selected points at other discharges, how well the selected points represent the verticals or cells, and how well the selected transects represent the stream.

In practical applications, it is important that the likely errors in model predications be estimated. This is typically done by "model validation," in which model predictions are compared with measured data different from those used to develop or calibrate the model. (Oreskes et al. (1994) have pointed out that this is not really validation, but we will use this common term for the process.) Lamouroux et al. (1995) present graphical comparisons of measured and predicted velocity distributions, although they acknowledge that their procedure is not strictly correct. Aceituno and Hampton (1988) compared the distributions of point measurements of depth and velocity separately with comparable distributions from PHABSIM verticals, but did not consider their joint distributions or estimates of WUA. Unfortunately, these examples are exceptions. Typically, validation is not even discussed, although validation for PHABSIM predictions seems particularly important; PHABSIM offers users a wide variety of options that can produce a wide range of results, so there is a danger that options may be selected consciously or unconsciously to produce a desired result (Bartz 1990; Gan and McMahon 1990).

The proper form of the validation will depend on the conceptual model underlying the PHABSIM modeling. As originally developed, the conceptual model for PHABSIM assumed that data from the transects applied half-way up or down stream to the next transect (Bovee 1982; Thomas and Bovee 1993). In other words, the stream is divided into horizontal cells, each of which is represented by measurements at one point on the transect. With this conceptual model, validation could simply involve measuring the depth, velocity and substrate at random points in the study reach at various discharges, and comparing these with the values assigned to the point by PHABSIM. It is important that the validation include the habitat variables and not just WUA, so that "correct" estimates of WUA that result from offsetting errors are revealed.

Recently, some PHABSIM users have used a different conceptual model in which transect data are treated as samples, stratified by habitat types, rather than as representing specific areas of the channel (e.g., CDFG 1991). The details of the validation would then depend on the details of the sampling scheme, but the basic process remains the same; model predictions of the joint distributions of depth, velocity, and substrate must be compared with independent data. Provided that transect sites are selected randomly, they would provide an unbiased estimate of conditions in the study reach, so that models could be validated at the transects, and the streamwise spatial sampling errors could be estimated separately using statistical methods such as bootstrapping (Williams 1996). Since the PHABSIM hydraulic models cannot be calibrated for the more turbulent areas of many streams, however, the condition of randomly located transects is difficult to meet in PHABSIM studies, and validating the model with data from randomly located points seems more appropriate. As with any statistic developed by sampling, estimates of WUA should be reported with standard errors or confidence intervals, so that decision makers are informed of the uncertainty associated with the estimates (Castleberry et al. 1996).

Application of Models to Aquatic Habitat

Since our discussion of habitat models is in the context of their application to evaluating habitat for a particular species and life stage of fish, the most relevant question is whether such models can capture aspects of the hydraulic environment that are most important to the organism in question. In some cases, the answer is clearly no. For example, chinook salmon select spawning sites on the basis of subsurface flow, as well as depth, velocity, and substrate (Healey 1991, Vyverberg et al. 1997), so a model that does not address subsurface flow will be seriously incomplete in its evaluation of habitat for spawning chinook salmon.

More generally, we argue that fish often respond to features in their hydraulic environments such as velocity gradients over small length scales. For example, trout may hold in the flow separation zone downstream of a boulder, as described for a Pennsylvania stream by Bachman (1984, p. 9):

> Typically, foraging sites were in front of submerged rocks, or on top of but on the downward-sloping rear surface of a rock. From there the fish had an unobstructed view of oncoming drift. While a wild brown trout was in such a site, its tail beat frequency was minimal, indicating that little effort was required to maintain a stationary position even though the current only millimeters overhead was as high as 60 to 70 cm/second. Most brown trout could be found in one of several such sites

day after day, and it was not uncommon to find a fish using many of the same sites for three consecutive years.

Contrast the precise positioning of this trout in the hydraulic environment (within millimeters of a steep vertical velocity gradient) with the detail that can be resolved in hydraulic models. Even with 2-Dflow models, the resolution is scaled by flow depths (Ghanem et al. 1996), and cannot account for vertical velocity gradients. The best that can be done is to patch on some estimated average velocity gradient, and as should be evident from Figure 2, this would give only a crude approximation. Accordingly, there is a discontinuity in the spatial scale at which it seems feasible to model the hydraulic environment, and the spatial scales at which fishes often respond to it. This seems particularly true for fishes that hold near steep velocity gradients, such as near the bed of the stream, boulders, or logs.

At best, practical modeling of the hydraulic environment for determining instream flows involves estimating the distributions or joint distributions of depth and velocity over sizable areas. Where the channel conditions are sufficiently uniform that this can be done with reasonable accuracy, this information would obviously be useful for thinking about the effects of discharge on fish habitat. If such information can be developed by mapping (Collings 1972) or by an empirical approach (Lamouroux 1995) it will be similarly useful. However, values of hydraulic variables averaged over sizable areas should not be confused with the local values to which fish and other organisms often respond (Railsback 1999; Bult et al. 1999). To combine hydraulic model results, which are accurate only on a coarse scale, with habitat preference or suitability data collected on a much finer scale, raises troubling questions about meaning. PHABSIM estimates of weighted usable area result, in effect, from multiplying biological apples by hydraulic oranges.

Railsback (1999) proposes dealing with this problem of scale mismatch by developing suitability data from observations in cells with a spatial scale comparable to the resolution of the hydraulic modeling. This raises another set of problems. If the cells are small then occupancy of each cell may be affected by occupancy of adjacent cells, as wells as by hydraulic factors, and collection of enough hydraulic data for modeling any sizable length of stream will be difficult and expensive. If the cells are large, then describing the cell by a single index for depth, velocity, and index is dubious, and the biological meaning of weighted usable area is compromised.

Conclusions

Flow fields in natural channels are complex, and it is not feasible to model this complexity for any length of channel at the finer length scales to which fish often respond. We believe that a more modest approach to using hydraulic models for instream flow assessments is appropriate. In many streams, 2-D modeling may produce reasonable estimates of the amount of habitat with given combinations of depth and average velocity, and in other streams this can probably be estimated empirically. This is important information that any of us would want to use if we were charged with making decisions about instream flows, if it can be obtained without taking up too much of the available funding.

We suggest, however, that it is prudent to leave the hydraulic and biological inquiries as separate and distinct tasks, in part because this helps avoid the appearance of models providing answers, rather than aids to thought. We suspect that the best way to evaluate the importance of hydrologic conditions for a particular fish is to have a good understanding of the way that the fish uses the hydraulic environment, the kind of understanding that is developed by careful observational studies such as Jenkins (1969), Bachman (1984), or Nielsen (1992), and especially from long-term studies such as those on Carnation Creek in British Columbia (Hartman et al. 1995), or Brows Beck in England (Elliott 1994).

Such evaluations involve use of professional judgement in considering data from hydraulic modeling or mapping, and can be criticized as subjective. However, modeling gives only an illusion of objectivity. Modeling always involves simplifying assumptions. Therefore, judgement goes into deciding just what and how to model, and good judgement requires knowledge of both the model and the thing being modeled. Models are not a substitute for knowledge and experience. Whether a model is good or bad depends upon the purpose to which it is put. For simulating depth and velocity, different models are appropriate for different kinds of channels and for different scales of resolution. However, all models have limitations. For simulating a particular reach of stream, the proper use of any model requires consideration of the statistical problems arising from sampling and measurement errors, and appropriate validation.

Acknowledgements

We thank Peter Whiting for providing us with the Solfatera Creek data and photograph, and Nicolas Lamouroux, Robert Jarrett, and Janice Fulford for helpful discussions of various topics.

References

Aceituno, M. and M. Hampton. 1988. Validation of habitat availability determinations by comparing field observations with hydraulic model (IFG-4) output. Pages 322-335 *in* K. Bovee and J.R. Zuboy, editors. Proceedings of a workshop on the development and evaluation of habitat suitability criteria. U.S. Fish and Wildlife Service Biological Report 88(11).

Bachman, R.A. 1984. Foraging behavior of free-ranging wild and hatchery brown trout in a stream. Transactions of the American Fisheries Society 113:1-32.

Bartz, B. 1990. Sources of uncertainty and effect on interpretation of results in the development of instream flows for fisheries habitat. Master's. thesis, Civil and Environmental Engineering, Utah State University, Logan, Utah.

Beebe, J.T. 1996. Fluid speed variability and the importance to managing fish habitat in rivers. Regulated Rivers: Research and Management 12:63-79.

Bovee, K.D. 1982. A guide to stream habitat analysis using the instream flow incremental methodology. Instream Flow Information Paper No. 12. U.S. Fish and Wildlife Service. FWS/OBS-82/26.

Bult, T.P., S.C. Riley, R.L. Haedrich, R.J. Gibson, and J. Heggenes. 1999. Density-dependent habitat selection by juvenile Atlantic salmon (*Salmo salar*) in experimental riverine habitats. Canadian Journal of Fisheries and Aquatic Sciences 56:1298-1306.

Carter, R.W., and I.E. Anderson. 1963. Accuracy of current meter measurements. Journal of the Hydraulics Division American Society of Civil Engineers. 89(HY4):105-115.

CDFG (California Department of Fish and Game). 1991. Instream flow requirements for brown trout, Rush Creek, Mono County, California. Stream Evaluation Report 91-2, California Department of Fish and Game, Sacramento.

Castleberry, D.T. and eleven others. 1996. Uncertainty and instream flow standards. Fisheries 21(8):20-21.

Collier, K.J. and Wakelin, M.D. Instream habitat use by blue duck (*Hymenolaimus alacorhynchos*) in a New Zealand River. Freshwater Biology 35:277-287.

Collings, M. 1972. A methodology for determining instream flow requirements for fish. Pages 72-86 *in* Proceedings of Instream Flow Methodology Workshop. Washington Department of Ecology. Olympia, Washington.

Dietrich, W.E. 1987. Mechanics of flow and sediment transport in river bends. Pages 179-227 *in* K. Richards, editor. River Channels Environment and Process. Basil Blackwell Ltd., New York.

Dietrich, W.E., and J.D. Smith. 1983. Influence of the point bar on flow through curved channels. Water Resources Research. 19(5):1173-1192.

Dingman, S.L. 1989. Probability distribution of velocity in natural channel cross sections. Water Resources Research 25:508-518.

Elliott, J.M. 1994. Quantitative ecology and the brown trout. Oxford University Press, London.

Fulford, J.M., K.G. Thibodeaux, and W.R. Kaehrle. 1994. Comparison of current meters used for stream gaging. Pages 376-385 *in* Fundamentals and Advancements in Hydraulic Measurements and Experimentation, 1994. ASCE Hydraulic Conference Proceedings.

Gan, K. and T. McMahon. 1990. Variability of results from the use of PHABSIM in estimating habitat area. Regulated Rivers: Research and Management 5:233-239.

Ghanem, A., P. Steffler, F. Hicks, and C. Katopodis. 1996. 2-D hydraulic simulation of physical conditions in flowing streams. Regulated Rivers: Research & Management. 12:185-200.

Greenberg, L., P. Svendsen, and A. Harby. 1994. Availability of microhabitats and their use by brown trout (*Salmo trutta*) and grayling (*Thymallus thymallus*) in the River Vojman, Sweden. Pages 606-624 *in* Proceedings of the 1st International Conference on Habitat Hydraulics.

Hartman, G.F., Scrivener, and M.J. Miles. 1995. Impacts of logging in Carnation Creek, a high-energy coastal stream in British Columbia, and their implications for restoring fish habitat. Canadian Journal of Fisheries and Aquatic Sciences 53(Suppl. 1):237-251.

Healey, M.C. 1991. Life history of chinook salmon. Pages 311-394 *in* Groot, C., and L. Margolis, editors. Pacific salmon life histories. University of British Columbia Press, Vancouver.

Heggenes, J. 1994. Physical habitat selection by brown trout (*Salmo trutta*) and young Atlantic salmon (*S. salar*) in spatially and temporally heterogeneous streams: implications for hydraulic modeling. Pages 12-30 *in* Proceedings of the 1st International Conference on Habitat Hydraulics.

Heggenes, J. 1996. Habitat selection by brown trout (*Salmo trutta*) and young Atlantic salmon (*S. salar*) in streams: static and dynamic hydraulic modeling. Regulated Rivers: Research and Management 12:155-169.

Herschy, R.W. 1978. The accuracy of current meter measurements. Proceedings of the Institution of Civil Engineers, Part 2: 65:431-437.

Jenkins, T.M. Jr. 1969. Social structure, position choice and microdistribution of two trout species (*Salmo trutta* and *Salmo gairdneri*) resident in mountain streams. Animal Behavior Monographs 2:56-123.

Jowett, I.G. 1989. River hydraulic and habitat simulation. RHYHABSIM computer manual. Ministry of Agriculture and Fisheries, New Zealand Fisheries Miscellaneous Report 49, Christchurch.

Lamouroux, N., Y. Souchon, and E. Herouin. 1995. Predicting velocity frequency distributions in stream reaches. Water Resources Research 31(9):2367-2375.

Lamouroux, N., H. Capra, and M. Pouilly. 1998. Predicting habitat suitability for lotic fish: linking statistical hydraulic models with multivariate habitat use models. Regulated Rivers: Research & Management 14:1-12.

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

Leclerk, M., A. Boudreault, J.A. Bechara, and G. Corfa. 1995. Two-dimensional hydrodynamic modeling: a neglected tool in the instream flow incremental methodology. Transactions of the American Fisheries Society 124:645-662.

Linsley, R.K., M.A. Kohler, and J.L.H. Paulhus. 1982. Hydrology for engineers. McGraw-Hill Co., New York.

Marchand, J.P., R.D. Jarrett, and L.L. Jones. 1984. Velocity profile, water-surface slope, and bed-material size for selected streams in Colorado. US Geological Survey Open-File Report 84-733, Lakewood, Colorado.

Marthur, D., W.H. Basson, E.J. Purdy, Jr., and C.A. Silver. A critique of the instream flow incremental methodology. Canadian Journal of Fisheries and Aquatic Sciences 42:825-831.

Milhous, R.T., D.L. Wegner, and T. Waddle. 1984. User's Guide to the Physical Habitat Simulation System (PHABSIM). Instream Flow Information Paper 11. U.S. Fish and Wildlife Service. FWS/OBS-81/43 Revised.

Milhous, R.T., M.A. Updike, and D.M. Schneider. 19989. Physical Habitat Simulation System (PHABSIM) Reference Manual-Version II. Instream Flow Information Paper No. 26. U.S. Fish and Wildlife Service. Biological Report 89(16).

Nielsen, J.L. 1992. Microhabitat-specific foraging behavior, diet, and growth of juvenile coho salmon. Transactions of the American Fisheries Society 121:617-634.

Oreskes, N., K. Shrader-Frechette, and K. Belitz. 1994. Verification, validation, and confirmation of numerical models in the Earth Sciences. Science 263:641-646.

Osborne, L.L., M.J. Wiley, and R.W. Latimore. 1988. Assessment of the water surface profile model: accuracy of predicted instream fish habitat conditions in low-gradient, warmwater streams. Regulated Rivers: Research and Management 2:619.631.

Petit, F. 1987. The relationship between shear stress and the shaping of the bed of a pebble-loaded river La Rulles - Ardenne. Catena 14:453-468.

Pouily, M., S. Valentin, H. Capra, V. Ginot, and Y. Souchon. 1995. Methode des microhabitats: priniples et protocoles d'application. Bulleltin Francais de la P^che et de la Pisciculture 336:41-54.

Railsback, S. 1999. Reducing uncertainties in instream flow studies. Fisheries 24(4):24-26.

Rantz, S.E. and others. 1982. Measurement and computation of streamflow: Volume 1. Measurement of stage and discharge. US Geological Survey Water-Supply Paper 2175, Washington DC.

Resier, D.W., T.A. Wesche, and C. Estes. 1989. Status of instream flow legislation and practices in North American. Fisheries 14(2):22-29.

Richter, B.D., J.V. Baumgartner, J. Powell, and D.P. Braun. 1996. A method for assessing hydrologic alteration within ecosystems. Conservation Biology 10:1163-1174.

Richeter, B.D., J.V. Baumgartner, R. Wigington, and D.P. Braun. 1997. How much water does a river need? Freshwater Biology 37:231-249.

Shirvell, C.S. 1986. Pitfalls of physical habitat simulation in the instream flow incremental methodology. Canadian Technical Report of Fisheries and Aquatic Sciences No. 1460.

Shirvell, C.S. 1994. Effect of changes in streamflow on the microhabitat use and movements of sympatric juvenile coho salmon (*Oncorhynchus kisutch*) and chinook salmon (*O.*

tshawystcha) in a natural stream. Canadian Journal of Fisheries and Aquatic Sciences 51:1644-1652.

Thomas, J.A. and K.D. Bovee. 1993. Application and testing of a procedure to evaluate the transferability of habitat suitability criteria. Regulated Rivers: Research and Management 8:285-294.

Townsend, F.W., and F.A. Blust. 1960. A comparison of stream velocity meters. Journal of the Hydraulics Division, American Society of Civil Engineers. 86(HY 4):11-19.

Vyverberg. K., B. Snider and R.G. Titus. 1997. Lower American River chinook salmon spawning habitat evaluation: October 1994. California Dept. of Fish and Game, Environmental Services Division, Sacramento.

Weiber, P.L. and J.D. Smith. 1987. Initial motion of coarse sediment in steep mountain streams of high gradient. Pages 299-308 *in* R.L. Beschta, T. Blinn, G.E. grant, G.G. Ice and F.J. Swanson, editors. Erosion and sedimentation in the Pacific Rim. International Association of Hydrological Sciences Publication No. 165.

Whiting, P.J. 1997. The effect of stage on flow and components of the local force balance. Earth Surface Processes and Landforms 22:

Whiting, P.J., and W.E. Dietrich. 1991. Convective accelerations and boundary shear stress over a channel bar. Water Resources Research. 27(5):783-796.

Williams, J.G. 1996. Lost in space: minimum confidence intervals for idealized PHABSIM studies. Transactions of the American Fisheries Society 125:458-465.

Table 1: Rules of thumb from Herschy (1978) for 95% confidence intervals for hydraulic measurements, expressed as % of measured values.

Uncertainty in or from:

current meter error	1% at 0.5 m/s, 2% at 0.25 m/s, 5% at 0.1 m/s
measurement of width	0.5%
measurement of depth	2.5%
time variation in velocity measurement	5% at 0.3 m/s, 22% at 0.1 m/s, 3 min. exposure
vertical spatial variation in velocity	7% (0.2 & 0.8 d method), 15% (0.6 d method)

Figure Legends

Figure 1. View of Solfatera Creek, looking downstream over the reach studied by Whiting and Dietrich (1991) and Whiting (1997). Note moderate gradient and apparently tranquil flow.

Figure 2: Downstream and cross-stream velocity fields at sections spaced 2 m apart in Solfatera Creek, Wyoming, reprinted from Whiting and Dietrich (1991). Isovels (lines of equal velocity) are at ten cm/s intervals; shaded areas indicate flow toward the left bank. Downstream isovels range from 0 to 70 cm/s, cross-stream isovels from 20 cm/s to the left to 30 cms to the right. The high velocity core near the bottom at Sections 1 and 2 (> 50 cm/s downstream) moves up and splits going over the bar in sections 7-10, with downstream velocity peaking at > 70cm/s in Sections 8 and 9. Velocity is highest near the right bank in Section 11 (> 60 cm/s), with a secondary maxima (> 50 cm/s) forming to the left of the bar. Water close to the right side of the bar in Section 11 is eddying upstream (< 0 cm/s). Section numbers increase in the downstream direction. See text for site description.

Figure 3. Estimates of the differences, in percentages, from estimating the vertically averaged water velocity using the velocity at 6/10s depth, and using the detailed measurements of Whiting and Dietrich (1991), for eight of the sections shown in Figure 1; the box plot summarizes the differences for all sections. Positive differences indicate that velocity at 6/10s depth is greater than the average estimated from the detailed measurements; each circle represents one vertical.

Figure 4 Standard deviations of velocity measurements averaged over different time periods, as percentages of the overall (one hour) means. Data from Carter and Anderson (1963); the anomaly in the 0.6 depth curve probably results from a typographical error.

Figure 5. Means and standard deviations of the differences between measured and modeled mean column velocities at verticals on the Williams Fork River, Colorado, for three PHABSIM hydraulic models (IFG-4, MANSQ, and WSP) calibrated at three discharges. Data from Table 4.5 in Bartz (1990).

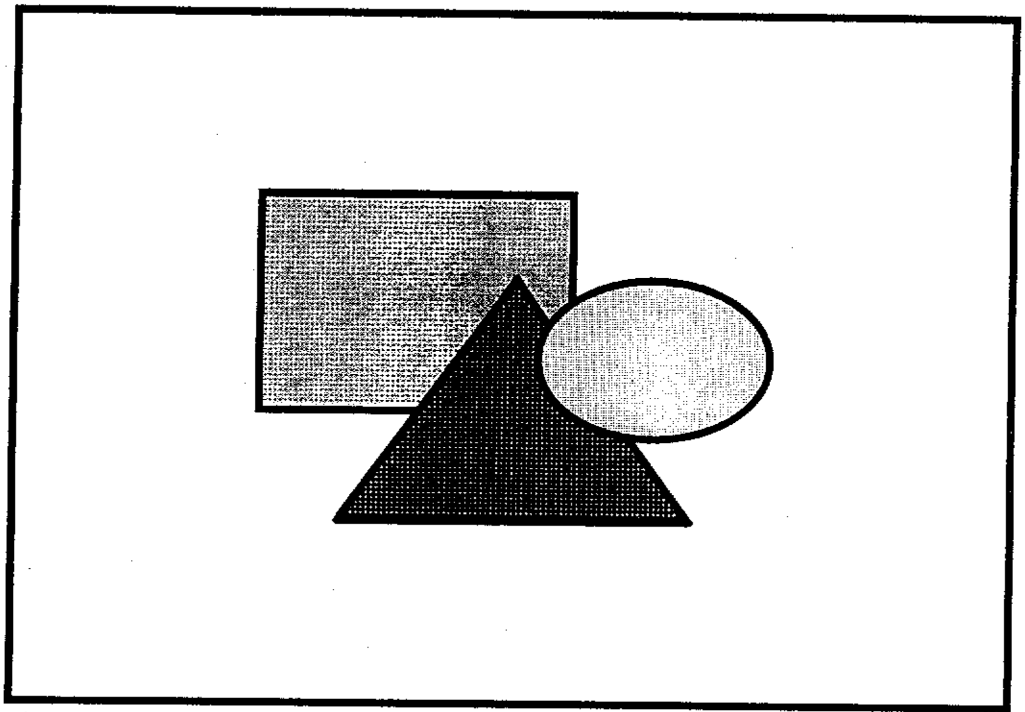
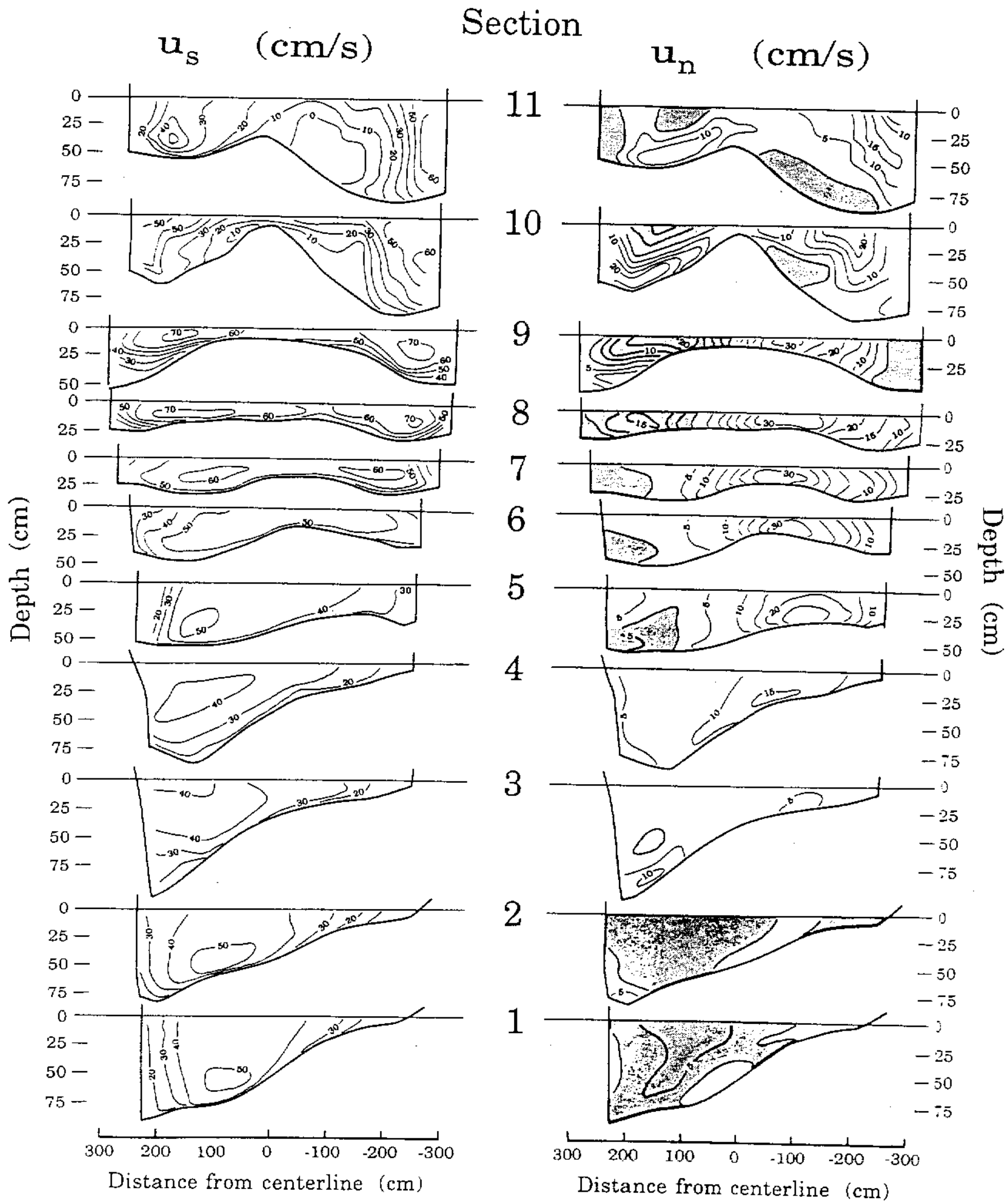


Figure 1



Section
u_s (cm/s)
u_n (cm/s)
11
10
9
8
7
6
5
4
3
2
1
Depth (cm)
Depth (cm)
300 200 100 0 -100 -200 -300
Distance from centerline (cm)
Distance from centerline (cm)

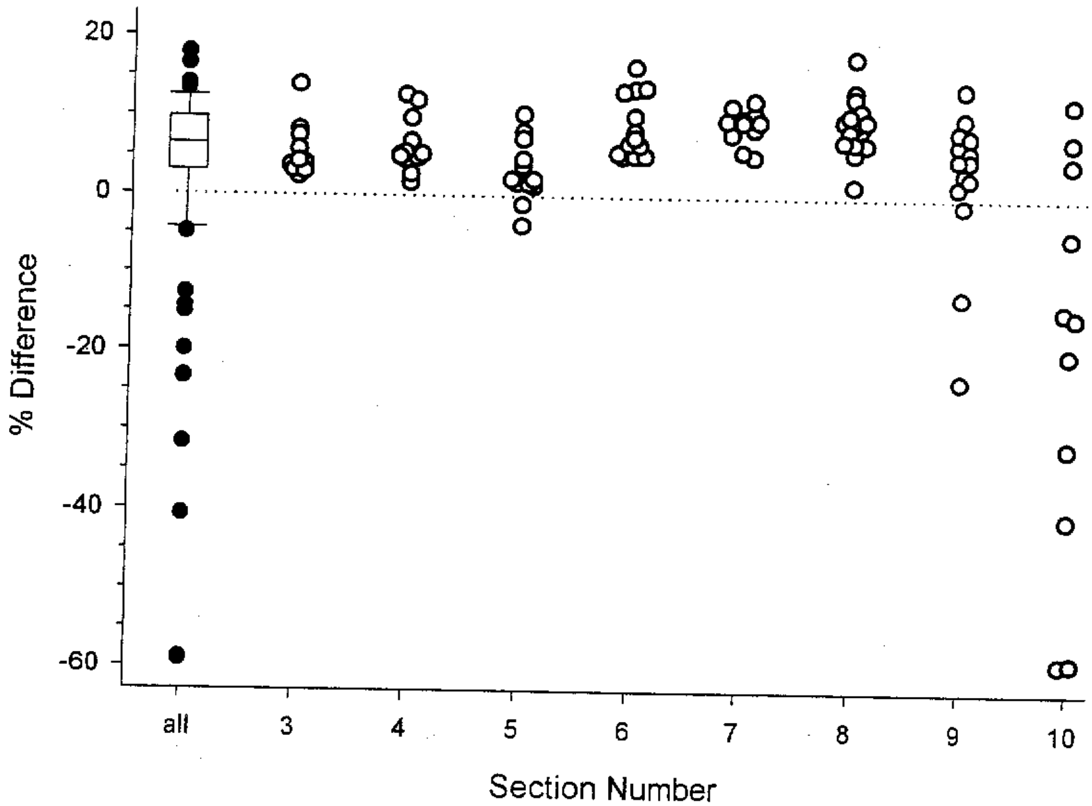
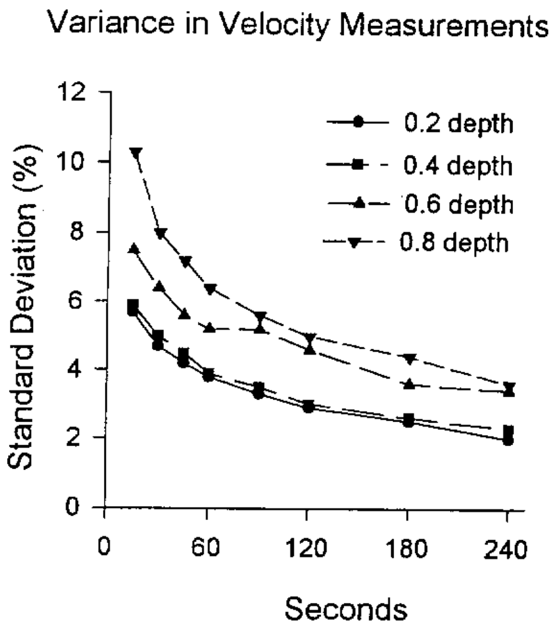
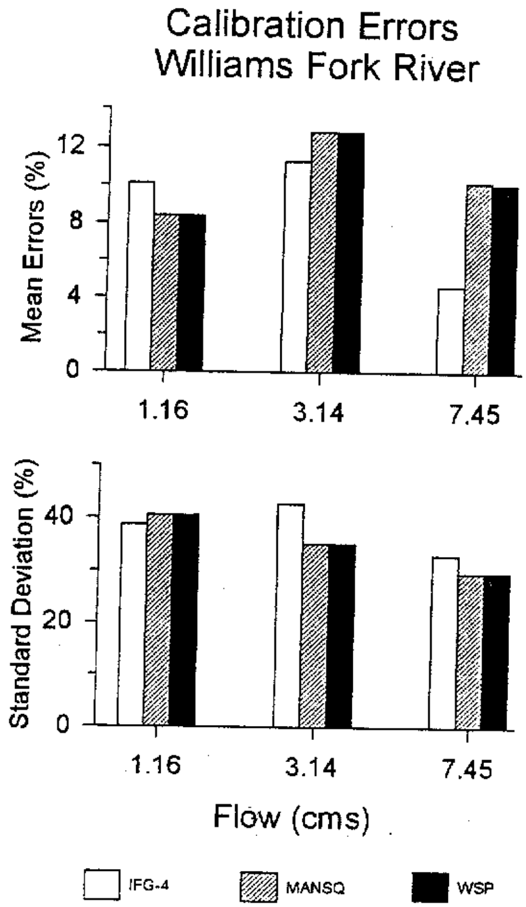


Figure 3


Variance in Velocity Measurements
Standard Deviation (%)
Seconds
0.2 depth
0.4 depth
0.6 depth
0.8 depth
0
2
4
6
8
10
12
0
60
120
180
240

Figue 4



Data from Bartz (1990)

Figure 5