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NOTES

A Critique of the Instream Flow Incremental Methodology

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A review and reanalysis of the published literature show that several assumptions are violated in the application of the Instream Flow Incremental Methodology (IFIM) without consideration of the implications of so doing. The fundamental assumption of a positive linear relationship between "potential available habitat" (WUA) and biomass of fish has neither been documented nor validated, particularly in warmwater streams. Absence of correlation precludes prediction of changes in fish populations. In some studies the test of this assumption appears to be equivalent to a calibration operation. The other assumption violated includes independent selection of habitat variables by fish. The presence of significant interaction among habitat variables can affect the stream flow recommendations. Another problem exists in application of Physical Habitat Simulation (PHABSIM): one WUA unit should not be interpreted as being equal to another in biological production or habitat value unless shown to be an exact replica. Several combinations of physical variables could give rise to the same amount of WUA, none of which may be correlated to the biomass of fish. The utilization, suitability, or preference curves should not be treated as probability functions; a rating of 1.0 is not equivalent to probability of 1.0. Care should be taken to distinguish between real behavioral preferences of fishes based on distributional occurrence from abundance (relative or absolute size) in a stream.

Un examen et une analyse des ouvrages publiés montrent que plusieurs hypothèses sont réfutées dans l'application de la méthode IFIM (« Instream Flow Incremental Methodology ») sans considération des répercussions. L'hypothèse fondamentale selon laquelle il existe une relation linéaire positive entre l'habitat potentiel disponible (WUA) et la biomasse des poissons n'a pas été étudiée ni prouvée, en particulier dans les cours d'eau chaude. L'absence de corrélation empêche la prédiction de variations dans les populations de poisson. Dans certaines études, la vérification de cette hypothèse semble être équivalente à un exercice d'étalonnage. Le choix indépendant par les poissons des variables de l'habitat constitue une autre réfutation de l'hypothèse. La présence d'une interaction importante entre les variables de l'habitat peut influer sur les recommandations en matière de débit. Il existe aussi un autre problème dans l'application de PHABSIM (« Physical Habitat Simulation ») : pour ce qui est de la production biologique et de la valeur de l'habitat, deux unités WUA ne doivent être considérées égales que si elles sont des copies exactes. Plusieurs combinaisons de variables physiques peuvent produire les mêmes valeurs WUA, dont aucune n'est peut-être en corrélation avec la biomasse des poissons. Les courbes décrivant l'utilisation, la convenance ont la préférence ne devraient pas être traitées comme des fonctions de probabilité : une évaluation s'élevant à 1,0 n'est pas équivalente à une probabilité de 1,0. La distinction entre les vraies préférences comportementales des poissons basées sur la fréquence de la répartition et l'abondance (relative ou absolue) dans un cours d'eau devra faire l'objet d'une attention particulière.

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The Instream Flow Incremental Methodology (IFIM) was developed by the U.S. Fish and Wildlife Service to provide a standard analytical technique for recommending flows for a stream. Originally, IFIM was developed for application in small simple cold-water stream systems from which water was to be diverted for off stream consumptive uses or to be allocated for other development projects. One of the initial objectives of the method was to assess with relative ease changes in fish standing crop and species composition due to changes in stream flow (Bovee 1978). Several assumptions are made in utilizing the method in a given stream. The purpose of our note is to critique some of these assumptions.

Only recently has the applicability of IFIM been investigated in a relatively small warmwater stream (Orth and Maughan 1982). These authors provided sufficient data in the published literature for our reanalysis. We will therefore use these data and other published examples for the evaluation of the underlying asssumptions. Undoubtedly, results of many other IFIM studies exist in unpublished reports, nonrefereed papers, proceedings, etc.

Theory and Mechanics of Calculations

An important integral component of IFIM is PHABSIM (Physical Habitat Simulation). The PHABSIM procedure contains four primary components: (1) physical measurements of depth, velocity, substrate, and cover within the stream reach, (2) computer simulation of the stream hydraulics, (3) determination of a composite probability of use from the suitability value for each combination of depth, velocity, and substrate found within the stream reach, for each species and life history phase, and (4) the calculation of weighted usable area (WUA) for each stream flow, species, and life history phase for each season. Other factors such as water quality and food can be also incorporated in the calculation of WUA but the level of complexity in application and interpretation increases substantially. Biological interactions (competition, predation, etc.) are recognized as important factors but at present cannot be included in the application of the method.

The outputs of computer simulations of physical habitat variables (based on measurements of depth, velocity, substrate, cover, etc.) in a stream reach and species life stage "probability of use" or "preference" or "suitability" curves (based on instantaneous fish measurements in the field, expert opinion, or from literature sources) are integrated into a potential available WUA for each flow. A relationship is then established between the availability of potential WUA for each life history phase of a species and stream flow in each season or time period.

Four basic assumptions are as follows: (1) depth, velocity, and substrate are the most important physical habitat variables affecting the distribution and abundance of fishes; behavioral preference of a life stage of a species for each physical variable can be established from instantaneous fish measurements in the field and probability of use curves or suitability or utilization indices constructed; (2) depth, velocity, and substrate independently influence habitat selection by fishes; (3) preference factors for depth, velocity, and substrate, etc., can be combined through multiplication to create WUA index; and (4) a positive linear relationship exists between size of WUA and the biomass of fishes; a slope of 1 is assumed to relate biomass and WUA (Bovee 1978). Presumably, an increase in the WUA will result in increased fish populations because populations are implicitly assumed to always be habitat limited.

The WUA (in optimum habitat equivalents), based on the composite suitability of three principal habitat variables, is derived for each stream reach from instantaneous measurements as follows:

$$\mathrm{WUA} = \sum_{i=1}^{n} C_i A_i$$

where $C_i = f_v(V_i) \times f_d(D_i) \times f_s(S_i)$, $f_v(V_i)$ = suitability weighting factor for the velocity in cell i, $f_d(D_i)$ = suitability weighting factor for the depth in cell i, $f_s(S_i)$ = suitability weighting factor for the substrate type in cell i, n = the number of cells, and A_i = the area of cell i. Weighting factors are derived from "utilization" or "preference" or "suitability," "electivity," or "probability of use" curves. The method assumes that behavioral characteristics of a life stage of a species can be defined by these curves. The mode or the peak of the curve is interpreted as the optimum value of a variable for fish usage and is given a weighting factor of 1. The tails of the curve represent zero weighting factor. Generally, weighting values between 0 and 1 are empirically determined (in equivalent optimum habitat units) from analysis of instantaneous observations of fish distribution over the range of each variable (Bovee 1982). An example of velocity and depth preference curve for adult smallmouth bass, *Salmoides dolomieui*, is given in Fig. 1.

Suitability Index and Preference Curves

The original application of IFIM, treatment of "suitability" or "preference" curves as probability functions, led to the calculation of a joint probability function by multiplication of univariate preference factors as simple conditional probabilities (Bovee 1978, 1982). This procedure is correct only when probabilities are statistically independent. A transformation of univariate preference factors into simple probabilities is erroneous. First, the mode or peak of curves shown in Fig. 1 only have a subjective rating of 1.0, which is not equivalent to a probability of 1.0. That is, the curves should not suggest that there is a 100% chance (a certainty) of locating a species population or specified segment of a population. A rating of 1.0 simply means that most organisms were observed or captured at that depth and/or velocity at the time of collection. The curve does not have any statistical distribution and cannot be considered as a probability function. The ordinate values between 0 and 1.0 (calculated from proportional scaling of fish catches) have been incorrectly interpreted as actual probabilities in PHABSIM. Probability is an area under the curve and not a value of the ordinate. We are not aware of any published study that has addressed these statistical or mathematical distinctions, and yet the suitability function in the form of the joint probability function continues to be used (Bovee 1982).

The ratings of the "preference" or "suitability" curves are ratios. However, these ratios are based upon a shifting denominator. For example, if largest number, 10, were obtained at a particular depth and/or velocity, these variables will be given a rating of 1.0. If in another sampling the greatest number was 100 organisms the same variables would also be given a rating of 1.0. Obviously, there is a difference in the biomass of 10 and 100 organisms. In our view the development of the "preference" curve as we described will lead one to expect low correlation between "suitability" and fish standing stock.

Because fishes may respond to a multitude of factors in the field, thus manifesting daily changes in their distribution, different curves may be obtained on different sampling dates or times within a season. For example, fishes change position from

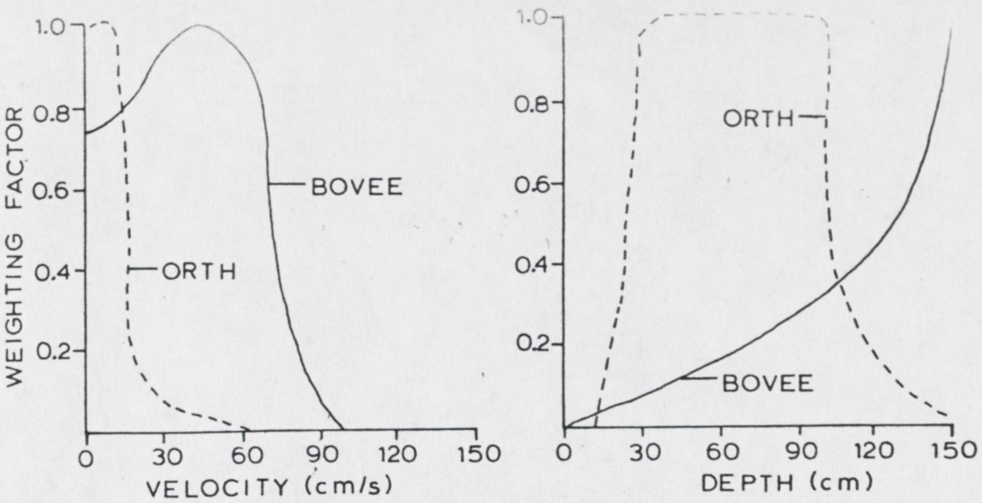

FIG. 1. Comparison of velocity and depth preference curves for the adult smallmouth bass from Bovee (1978) (solid lines) and Orth (1980) (broken lines). Orth used the term weighting factor, and Bovee used probability.

day to day, day versus night, and in response to prey–predators, etc. (Emery 1973; Helfman 1981; Fraser and Emmons 1984). Yet the total number of fish in a stream usually does not change over the period of observations on these normal behavioral responses. It is important for investigators, therefore, to distinguish between distribution (occurrence of a life stage at a location) and abundance (absolute or relative size) of a fish population at a stream reach.

A careful evaluation of the usability of a "preference" or "suitability" curve is necessary. The shape of the curve can be highly dependent on the site, stream, and time of collection. Consider the velocity and depth suitability curves developed using a similar method for the adult smallmouth bass from two different sources in Fig. 1. No resemblance is evident, and the optimum "preferred" or "suitable" depth and velocity differ by an order of magnitude between curves for each variable.

A restriction placed in IFIM on the development of preference curves is that they be obtained from unexploited streams at carrying capacity (Bovee 1982). Such conditions are rare, at best, especially for high-quality habitats. It is also rare that before initiating a study that the investigators know the carrying capacity of streams or that of a given stretch within a stream. Note that only when a substantial positive correlation is obtained between WUA and fish biomass a posteriori is the fish population considered to be at carrying capacity (Orth and Maughan 1982).

Assumption of Independence

The derivation of WUA is based on the assumption that organisms select each habitat variable independently of the other variable(s). Examination of the published analysis shows violation of this assumption. In table 3 of Orth and Maughan (1982) the interaction term for depth × velocity was highly significant ($P < 0.01$) in three of the four analyses presented. Interaction exists where two factors combine to produce an added effect not due to one of them alone; neither factor individually can show the "best" predictor to optimize the habitat. That is, the suitability of a given depth may depend upon velocity. For example, in the case of central stoneroller, *Campostoma anomalum*, the interaction of depth and velocity explained about 30% of the *known variation* in abundance of fish during spring. None of the other variables were significant. Because the two variables (depth and velocity) are highly interactive they may not be used to calculate unbiased estimates of WUA under the assumption of independence.

Orth and Maughan (1982) believed this interaction would be of limited significance on their minimum flow recommendations; a preference for greater depths at greater velocities would be observed. Using the exponential multiple regression models given by Orth and Maughan (1982) in their table 4, we estimated the response surface for densities of adult central stoneroller at various depths and velocities (Fig. 2). The effects of the depth–velocity interaction are obvious; fish densities increase radically with a slight increase in either depth or velocity. High densities of central stoneroller will occur (according to the model) at very low velocities and shallow depths. In fact, this model suggests that highest densities will occur at no flow, an illogical conclusion that can result when assumption of independent selection of variables is not met. The violation of the assumption of independence may be more universal than might have been previously believed. Studies by Gore and Judy (1981) and Orth and Maughan (1983) on benthic organisms also show the violation of assumption of independence. At any rate, because depth and velocity are correlated in a stream (Fraser 1972), it is difficult to meaningfully ascertain in the field that the selection of one habitat variable by fish is independent of the other variable.

The necessity of the assumption of independent selection of the habitat variables, of course, disappears when "suitability" or "preference" or "utilization" is not considered equivalent to probability in calculation of WUA. In fact, the composite WUA index need not be assumed to have any particular statistical property. It should be considered merely as a derived variable obtained from multiplying the weighting factors. The weighting factors can be mathematically manipulated in other ways with equal justification. That is, these factors can be added, or be treated logarithmically. The only relevant tests are as follows: Is WUA positively related to fish biomass in a predictable manner; is this relationship stable; and is this relationship robust?

Assumption of Positive Linear Relationship of WUA and Standing Crop

The basic foundation of IFIM is the calculation of WUA at incremental flows and assumes a positive linear relationship

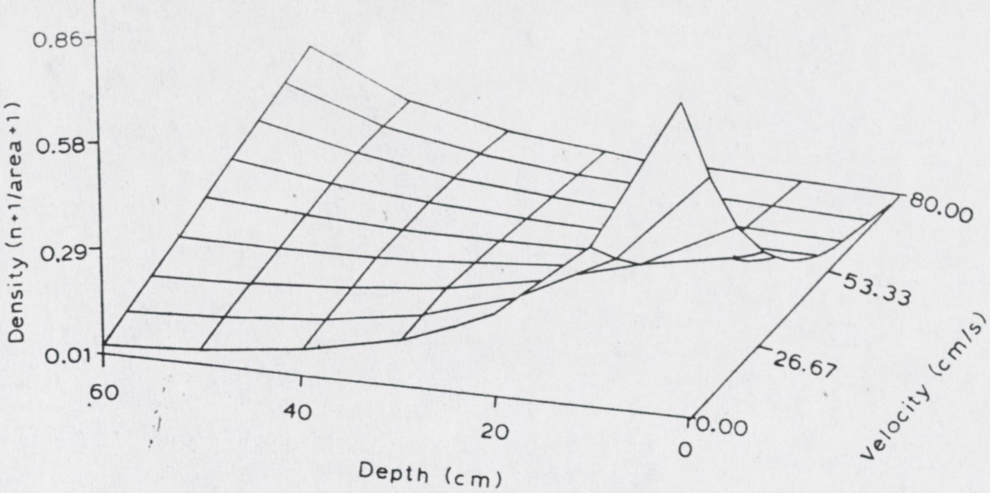

FIG. 2. Predicted response surface for abundance of the central stoneroller in Glover Creek, Oklahoma. The exponential model of Orth and Maughan (1982) in their table 4 was used for prediction.

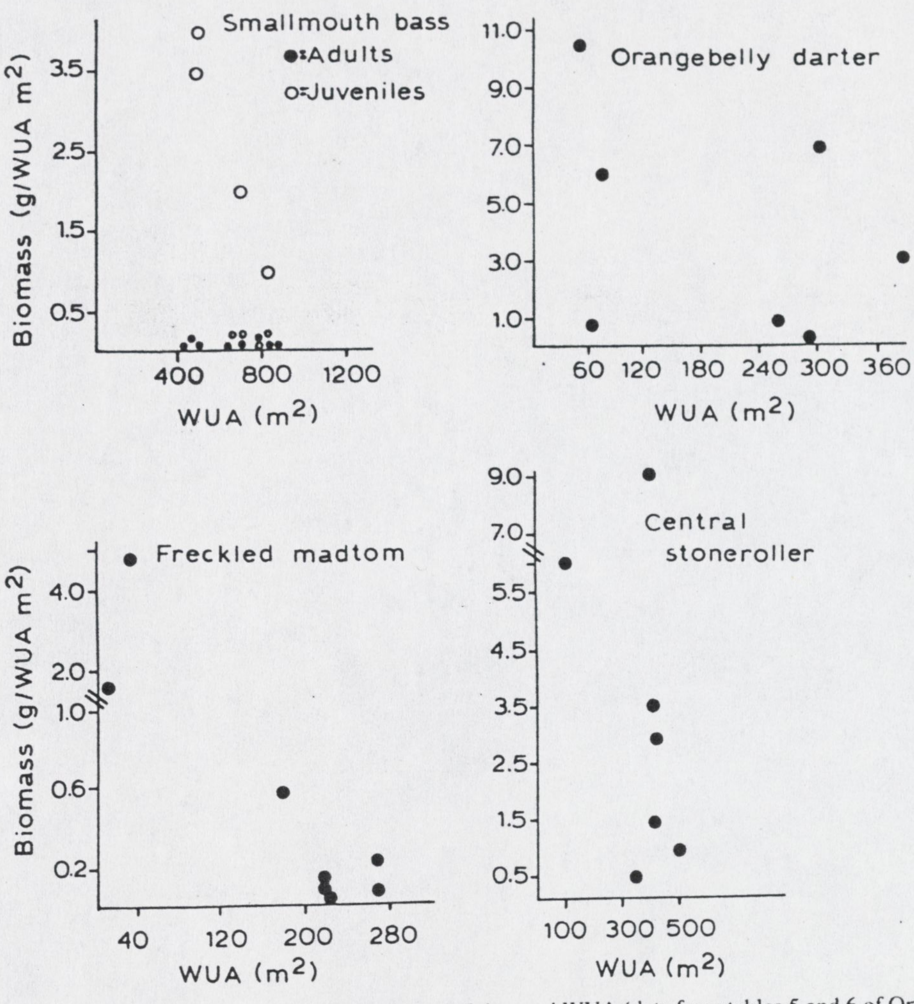

FIG. 3. Plots of standing stock of Glover Creek fishes and WUA (data from tables 5 and 6 of Orth and Maughan 1982).

between WUA and standing stock of fish (grams). The regression line for this relationship should pass through the origin, that is, no habitat no fish. Yet these assumptions are treated lightly in the instream flow assessments and subsequent recommendations. The utility of a model depends on the degree to which it is predictive. If a strong relationship exists, the predictions can be made and verified on the changes in fish stoc with the corresponding alteration in flow regimes. No validatic of this relationship, although necessary, has been documente in the primary literature.

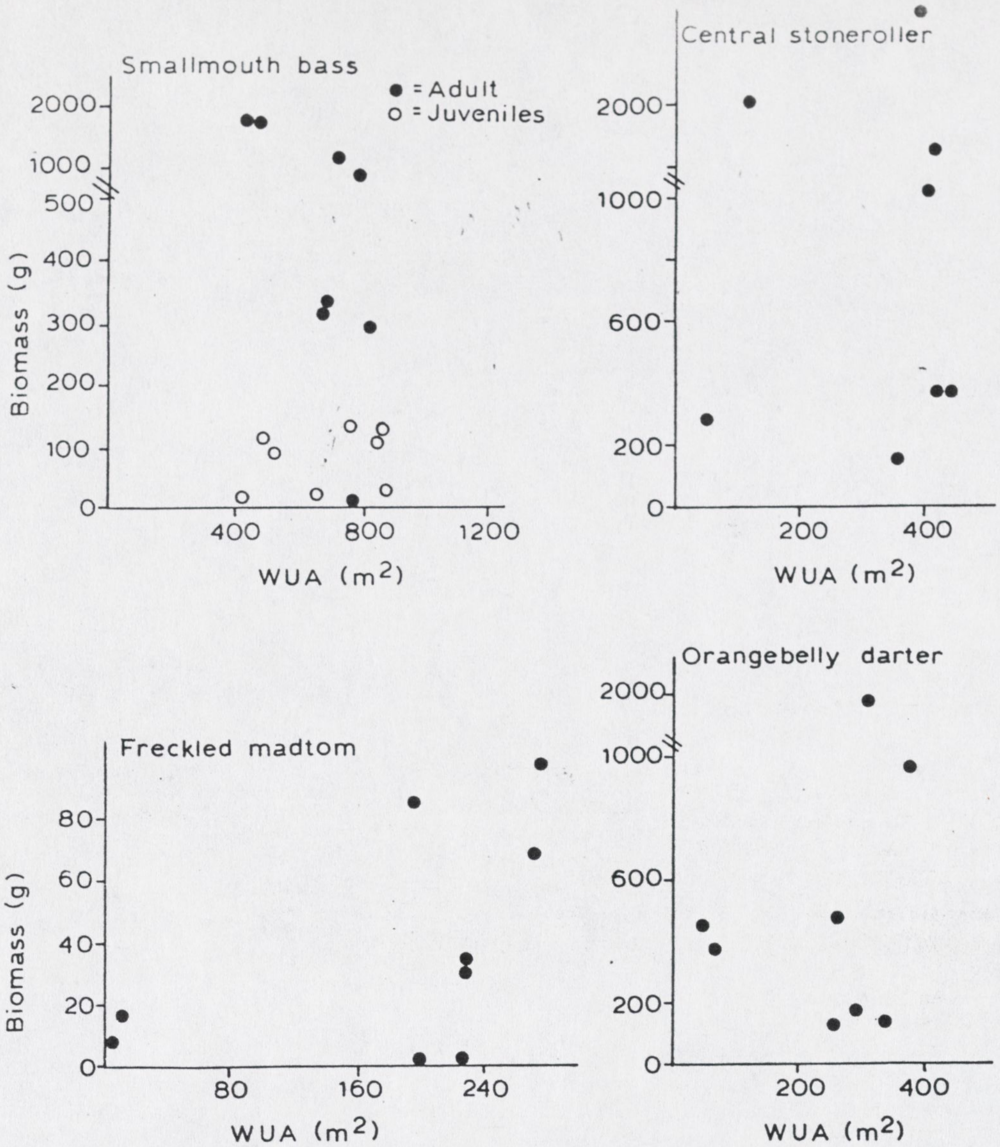

FIG. 4. Plots of standing stock of Glover Creek fishes and WUA (data from tables 5 and 6 of Orth and Maughan 1982).

The presence of a significant positive correlation between the WUA and biomass of fishes (grams) is interpreted as confirmation that physical habitat limits the fish stock and the population is at carrying capacity (Orth and Maughan 1982): The absence of statistical correlation is interpreted that factors other than those measured limit the standing stock. We have plotted in Fig. 3 and 4 the seasonal fish biomass and WUA data for smallmouth bass, orangebelly darter (*Etheostoma radiosum*), freckled madtom (*Noturus nocturnus*), and central stoneroller given by Orth and Maughan (1982) in their tables 5 and 6 to examine the assumption of linear positive relationship (grams vs. WUA). Both measures of biomass (grams and grams per WUA) are highly variable and a positive relationship is not apparent (Fig. 3 and 4). In fact, the trend is often negative, particularly that of the grams per WUA vs. WUA relationship. In addition, the lines do not appear to pass through the origin.

Some IFIM studies (Gore and Judy 1981; Orth and Maughan 1983) on benthic organisms have tested the above assumed linear relationship by transforming the original biomass data to $\ln(X + 1)$. These transformed data were then repressed on joint preference values derived from the raw biomass data (Gore and Judy 1981; Orth and Maughan 1983). Despite this calibration operation, i.e. regression of the original dependent (derived) variable, preference factor, against the original independent variable (biomass), consistent strong predictive relationships were absent. This result is surprising, since the preference or suitability values have little meaning except in terms of biomass or densities.

The slope of the relationship between fish biomass (kilograms) and WUA is kilograms per WUA and therefore should be a constant, particularly for a population at carrying capacity in a given stream. This conclusion is merely an extension of the IFIM original assumption of a slope of 1 (Bovee 1978) for the relationship between biomass (kilograms) and WUA for fish populations at carrying capacity. That is, a unit change in WUA results in a unit change in biomass. In other words, 15 units of composite WUA should support the same amount of fish biomass as 15 units elsewhere in the stream. The severalfold variability observed in Fig. 3 should not occur.

The calculations of WUA implicitly consider each habitat unit as biologically equivalent (Bovee 1982). However, there is no reason to believe that habitat units as defined by PHABSIM

TABLE 1. Distribution of WUA (m^2/km) for adult smallmouth bass at various depth–velocity combinations (data extracted from table 1 of Bovee 1978).

	Velocity (m/s)		
Depth (m)	0.15–0.30	0.30–0.45	0.45–0.60
0.30–0.45	15	—	14
0.45–0.60	17	15	21
0.60–0.75	17	15	6
0.75–0.90	11	44	63
0.90–1.05	15	15	9

are biologically equal and provide similar production rates unless each is an *exact replica* of the other unit. If each unit of WUA is biologically identical, a prediction of standing stock of fishes as a result of flow alteration may be possible. However, in the present form of calculations in PHABSIM, several combinations of depth, velocity, and substrate can give the same amount of WUA, none of which may support a similar fish biomass. We have extracted data from Bovee (1978) in Table 1 to show that at least five different combinations of depth and velocity provide the same amount of WUA (15 m^2/km) for adult smallmouth bass in a stream. A lack of correlation between WUA and fish biomass may in part result from treating each habitat unit as equivalent of another.

Effects of Outliers on Relationship of WUA and Biomass

The relationship between fish biomass (kilograms per hectare) and WUA (expressed as percent of total area) is used in some investigations. It is also weak and of questionable interpretation. For example, of the 20 correlations presented by Orth and Maughan (1982), 14 (70%) were not significant ($P > 0.05$). Examination of these reported relationships shows that in most cases only two or three *effective* data points existed (a cluster of points near the origin and one or two extreme values). A cluster of points is equivalent to only one *effective* data point in the determination of the slope of the regression line. All data points should weigh equally in determining a regression line (Draper and Smith 1979). Spurious correlations and therefore questionable conclusions can result when regressions are based on only two or three *effective* data points.

The following examples will illustrate the influence of outliers on habitat relationships and subsequent interpretations. Removal of single outliers from the central stoneroller summer ($X = 42.0$, $Y = 51.8$) and fall ($X = 42.9$, $Y = 17.7$) data sets reduced the reported significant correlations ($P < 0.01$) from 0.835 to 0.61 for the summer and from 0.737 to 0.416 for the fall; both correlations became nonsignificant ($P > 0.05$). We therefore consider the original interpretation that physical habitat limits population of central stoneroller to be suspect.

On the other hand, removal of outliers from the spring central stoneroller data sets ($X = 47.2, 46.0$, $Y = 0.9, 1.1$) increased the correlation from 0.159 ($P > 0.05$) to 0.892 ($P < 0.01$). Note that these are high values of WUA associated with low values of biomass (kilograms per hectare) and that their removal in no way biases the relationships that are assumed in IFIM. Using the same logic the original interpretation that the physical habitat did not limit the central stoneroller population becomes erroneous. Similarly, the removal of a single outlier ($X = 70.9$, $Y = 0$) from the adult smallmouth bass summer data set increased the reported correlation from 0.423 ($P > 0.05$) to 0.857 ($P < 0.01$). In the cases above, one or two data points explained over 50% of the variation in the data. Since no similarity exists in the configuration of the data points between years or season, we interpret these data to represent seasonal or yearly natural variations. A large portion of the variability observed here, perhaps, reflects the independent nature of the changes in WUA and fish populations. In the absence of such basic linear positive relationships no mechanism exists that can separate the natural variations in abundance of fishes from those due to changes in stream flow, nor can the model be validated.

Validation of Fish Biomass and Habitat Relationship

One of the reported strengths of IFIM via PHABSIM is its ability to assess changes in standing crop of fishes due to changes in flow regimes (Bovee 1978). Therefore, the question is would a short-term denial or loss of physical space or "habitat" result in a corresponding reduction of fish populations in streams? A non-IFIM designed experimental study conducted by Kraft (1972) provides some insights to the above question. He studied the effects of experimentally induced flow reductions on a brook trout, *Salvelinus fontinalis*, population in a simple cold-water stream in Montana. No declines in total abundance of fish occurred when the summer flow was reduced from 1.0 to 0.2 m^3/s (an 80% reduction in flow) for 90 d. A redistribution of brook trout was observed. Examination of his data for an experimental 90% flow reduction for 90 d, after accounting for natural fish losses at control stations and egress of fish out of the test sections, did not show a significant change in the standing stock of brook trout either. In fact, both the number and weight of trout in pools of the test sections of the stream increased, while comparable values at the control sections decreased, clearly suggesting redistribution of fish and not a true change in abundance. Therefore, temporary distributional changes may not result in corresponding changes in total abundance of species life stage and should not be interpreted as such. The study also indicates, at the least, that the availability of usable area may not have an immediate regulatory effect on the population. In application of IFIM, however, it is implicitly assumed that such changes will occur in fish populations as a result of flow alterations. In our view, investigators should attempt to relate changes in physical habitat in terms of their effects on the standing stock (abundance) in a stream as a whole rather than in temporary redistribution of a segment of the fish population in a stretch of a stream.

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Comparison of the Phosphorus–Chlorophyll Relationships in Mixed and Stratified Lakes

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Riley, E. T., and E. E. Prepas. 1985. Comparison of the phosphorus–chlorophyll relationships in mixed and stratified lakes. Can. J. Fish. Aquat. Sci. 42: 831–835.

Data from the literature were used to calculate separate regressions of summer chlorophyll *a* concentration ([Chl *a*]) on spring total phosphorus concentration ([TP]) for lakes that remain thermally stratified during the summer and lakes that mix intermittently during the summer. Significant differences were found in the spring [TP] – summer [Chl *a*] relationships for the two lake types ($P < 0.05$). The mean ratios of summer [TP] to spring [TP] were also significantly different in stratified and mixed lakes ($P < 0.001$); this difference is the explanation offered for why the spring [TP] – summer [Chl *a*] relationships were different in stratified and mixed lakes.

Des données tirées d'ouvrages publiés ont été utilisées pour calculer différentes régressions entre la concentration estivale de chlorophylle *a* ([Chl *a*]) et la concentration totale printanière de phosphore total ([PT]) dans des lacs qui restent thermiquement stratifiés pendant l'été et d'autres lacs qui se mélangent de façon intermittente en été. Des différences significatives ont été découvertes dans les relations [PT] au printemps et [Chl *a*] en été dans les deux genres de lac ($P < 0{,}05$). Les rapports moyens entre la [PT] en été et la [PT] au printemps étaient aussi significativement différents dans les lacs stratifiés et les lacs mélangés ($P < 0{,}001$) ; cette différence sert à expliquer la différence dans les relations [PT] au printemps – [Chl *a*] en été, dans les lacs stratifiés et mélangés.

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Total phosphorus (TP) is a good predictor of the standing crop of phytoplankton in lakes (Sakamoto 1966; Dillon and Rigler 1974; Nicholls and Dillon 1978; Oglesby and Schaffner 1978; Smith 1982; Prepas and Trew 1983). Generally, spring [TP] is the parameter used to predict the summer chlorophyll *a* concentration ([Chl *a*]). However, a substantial error is associated with predictions from these models. To reduce this error, authors have used variables other than spring [TP] as a predictor and [Chl *a*] as an indicator of phytoplankton standing crop. Nicholls and Dillon (1978) showed less error associated with predictions of algal biomass than with summer [Chl *a*]. Smith (1982) incorporated total nitrogen concentration ([TN]) along with [TP] as a predictor of summer [Chl *a*]. Prepas and Trew (1983) found that summer [TP] was a better predictor of summer [Chl *a*] than spring [TP]. However, spring [TP] and summer [Chl *a*] are still the most useful variables in these models, since spring [TP] can be predicted from loading models (Vollenweider and Kerekes 1980) and [Chl *a*] and spring [TP] are much easier to measure than phytoplankton biomass and summer [TP].

One source of variation in the spring [TP] – summer [Chl *a*] relationship that has been proposed, but not thoroughly tested for, is the influence of mixing patterns. It has been suggested that mixed lakes (i.e. lakes that mix intermittently during the summer) produce more summer Chl *a* per unit of spring TP than stratified lakes (i.e. lakes that remain thermally stratified during the summer). Oglesby and Schaffner (1975) assumed that the winter [TP] – summer [Chl *a*] relationships are different in the two lake types and calculated separate linear regressions of summer [Chl *a*] on winter [TP] for stratified and mixed lakes. Although there was generally more Chl *a* per unit TP in the

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Establishing Instream Flow Needs in Minnesota

by

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Minnesota is well known for its lakes and streams, extensive underground water, natural areas, fish and wildlife, and scenic beauty. The State's natural heritage is preserved by the most protective natural resources laws in the Nation. Minnesota law affords many opportunities for protecting instream uses, including: establishment of protected flows where water withdrawals threaten instream uses; protection of river corridors through wild and scenic river and critical area programs; review of public and private activities affecting the environment; and the right of all citizens to go to court to protect natural resources and environmental values. The State's natural resources programs are administered by the Minnesota Department of Natural Resources.

*Hedia Rieke is the Supervisor of the Water Allocation Unit, Division of Waters, Minnesota Department of Natural Resources.

In 1937, Minnesota established a system which requires a permit from the Department of Natural resources before surface or ground water can be withdrawn. Small domestic uses are exempted. In Minnesota, use of water must conform to the "riparian reasonable use" rule, which balances the interests of the users, of other riparian landowners, and of the public. Conflicts between users, referred to as appropriators, are not resolved on the basis of "first-in-time is first-in-right" as under the appropriation doctrine. Instead, a system of "priorities" has been established, with domestic water supply first; consumptive use of fewer than 10,000 gallons per day, second; agricultural irrigation and processing, third; power production, fourth; and all other uses, fifth. Appropriators with a lower priority may be required to curtail withdrawals for the benefit of higher priority users.

Permits issued for appropriation of water from streams or lakes are limited in order to maintain and protect instream uses; withdrawals for consumptive purposes are not allowed during periods when flows or water levels are below protected flows or protection elevations. "Protected flows" or instream flow needs refer to the amount of water required in a stream to maintain instream values, such as water-based recreation, navigation, aesthetics, fish and wildlife, and water quality.

When the law was passed in 1977 allowing for the establishment of protected flows, the Department was under considerable pressure to issue permits from surface water sources (drought 1976-77). Decisions on establishing protected flows have been made with less than adequate information regarding the impact on instream values. Instream flow assessments arrived at a single value stream flow--a minimum flow requirement for fishery resource. Such instream flows were often determined solely from an analysis of hydrologic records, (low-flow frequency analysis) providing limited opportunity for negotiation for fish maintenance and preservation. By 1978, minimum flows had been established for over 36 Minnesota rivers.

Contents

In 1980, the Department adopted regulations for water appropriation. Factors which must be considered when establishing protected flows were identified. They are: watershed characteristics, flow regime, river physical characteristics, aquatic system, riparian vegetation, water quality, existing fish and wildlife management, and alternative sources of water supply. These new requirements placed pressure on instream use advocates and water managers to identify instream values, to develop improved data collection and analytical techniques for determining instream flow needs, and to provide substantive instream flow input into the decision making process regarding allocation of water resources.

Having a strong legal framework for protecting instream flow needs is necessary but not sufficient. We still have to deal with the problem of determining how to actually protect aquatic life and make fair resource allocation decisions. Considerations in developing instream methods that are applicable to Minnesota are: species diversity; geographic variations of the biological and hydrologic data; and extensive recreational resources.

With respect to species diversity, Minnesota contains several major watersheds whose aquatic species vary significantly between northern and southern watersheds. Some streams, such as the Mississippi, support a large number of species, many of which are desirable game species. Objective decision making on determining the most critical species is very important. The use of simulation models may help decision makers determine the amount of suitable habitat that will be available for a given flow, the premise being that water appropriation decisions should not allow flows which will jeopardize a fish population. However, even though the simulation approach is less subjective than others, it is not without problems with respect to its application to Minnesota. The complex conditions (i.e., unstable channels, large bed streams, divided flows) found in many of the State's rivers may limit application of simulation models. A significant amount of data on numerous reaches of the river and the necessary expertise make the models costly and lengthy to apply.

Besides the problem of making decision methods developed elsewhere sensitive and applicable to the variation in conditions found in the State, there are other problems: problems of minimizing the effect of emotional interests in the process of establishing instream flow requirement; problems of defining in exact and precise terms the decision making criteria; and problems of adhering to the principles of good resource management decision making (i.e., simplicity) in developing an acceptable method for allocating water resources.

Indeed the success or failure of establishing protected flows at a desired level depends upon how well the need and justification for such flows are presented to decision makers, affected water users, and the public interest involved.

What is hard for a decision maker and even harder for appropriators to believe and recognize is the argument by instream use advocates that failure to maintain the desired flow will cause the resource to be adversely affected. If it can be shown that a reduction of the flow by "x" amount will result in "y" amount reduction in the resource, it will be understood and may even be supported. Such an approach allows the decision-makers to evaluate the effect of not maintaining the flow for instream uses at certain levels.

We also need to know when drought conditions occur, what percentage reduction can be applied to instream flows for short periods of time without seriously affecting the resource. As decision makers, we are used to these considerations when making decisions on allocation. However, for fishery and recreational interests, this may mean compromising the resource.

CONCLUSION

Decision making regarding instream flow needs has to consider complex legal and technical factors. Decisions are made only after considering and comparing beneficial and adverse effects and laws and regulations that apply to water allocation. Our decisions can be challenged through the administrative procedures and in courts. We are becoming aware of the need to base our decisions on sound information and analysis. Also presentation of arguments on the recommended flows has to be reasonable, credible and supported.

If instream flow needs are to be protected and accepted by water users, we must be able to demonstrate the need to maintain the flow at certain levels during various flow conditions, as well as document the effect of not obtaining those flows. We also need to be open to short-term compromises when they will not adversely affect the resource. Let us not forget the importance of applying the concept of administrative simplicity in making resource allocation decisions.

Technical Notes

The Physical Habitat Simulation System
by
Robert T. Milhous

Passage Habitat[1]

The passage habitat at a given streamflow is determined by the minimum of the passage habitats in the individual stream cross sections; consequently, the individual cross sections have to be considered instead of the total reach. In order to use this approach, passage criteria must be developed in terms of velocities and depths. An approach to determining the passage habitat is explained in PHABSIM Technical Note #2.

[1]PHABSIM Technical Note #2: Calculation of Passage Habitat, Instream Flow Group, October 1984.

The Divided Flow Approach to Simulating Physical Habitat in Reaches with Islands[2]

The physical habitat in a stream with large islands can be simulated using PHABSIM by treating each channel in a separate simulation. The total physical habitat is determined by the sum of the habitat in the separate channels. If the individual channels are treated as separate streams, the field work must be done with that objective in mind. In this case, the most important new data needed is the relationship between the flow in the channel and the total flow of the stream.

[2]PHABSIM Technical Note #3: The Simulation of Physical Habitat in a Reach with Islands, Instream Flow Group, October 1984.

Instream Flow Group

Back row, L to R: Bruce Wahle -Aquatic Biologist/Consultant
Ken Bovee - Hydrologist
Terry Waddle - Hydrologist
Lee Lamb - Management Analyst
Bob Milhous - Hydrologist
Dick Fisher - Ecologist
Johnie Crance - Fishery Biologist
Third row, L to R: Lisa Benson - CSU Work Study Student
Madeline Sieverin - Editorial Assistant
Connie Snelling - Secretary
Diane Schneider - Technical Services
Second row, L to R: Eve Schauer - Hydrologist/Consultant
Keith Garlinghouse - Engineering Aid/CSU Coop. Student
Cindy Collins - Hydraulic Engineer/Consultant
Kate Brandes - Editor (CSU)
Clair Stalnaker - Group Leader
Front row, L to R: Paula Seno - CSU Work Study Student
Leah Wilds -CSU Coop. Ed. Grad. Student
Pat Nelson - Fishery Biologist/Consultant
not in picture: John Bartholow - Wildlife Biologist

IFIM IS AN ISSUE BEFORE FERC

by
Edward F. Lawson*

The suitability of the Instream Flow Incremental Methodology (IFIM) as a methodology to resolve the question of minimum flows at the Conowingo Dam (Project No 405) on the Susquehanna River is now before the Federal Energy Regulatory Commission (FERC).

In renewing the license for the Conowingo project, the FERC ordered the licensees to consult with the intervenors (including the States of Pennsylvania and Maryland and the Susquehanna River Basin Commission) and agree on a study to determine the appropriate level of minimum flows. Operation of the dam in a peaking mode has resulted in dewatering of substantial areas of habitat, depressed levels of D.O and caused fish kills. The licensees favored a long-term population based study and would not agree to conduct an IFIM study as requested by intervenors. As a result, the issue was set for an adjudicatory hearing before a FERC Administrative Law Judge.

During the hearing, it became clear that there were a number of deficiencies with the licensees' population-based approach. These included: (1) a lack of baseline data; (2) a failure to specify predictive models; (3) an inadequate number of low regimes; (4) excessive duration and costs; and (5) lack of demonstrated utility for decision-making. Ken Bovee of the Instream Flow Group (IFG) was the expert witness on IFIM for the intervenors. His testimony demonstrated that an IFIM study could be conducted on a large warm-water river at a reasonable cost. Moreover, Mr. Bovee demonstrated the advantages of IFIM in making a decision on minimum flows including the methodology's ability to: (1) provide an analysis of the type of habitat limiting population; (2) determine the effect of a particular flow on habitat; (3) evaluate the impacts of high flows; (4) evaluate conflicts among different species and life stages; and (5) objectively select a flow for subsequent monitoring.

In spite of the evidentiary record, the Administrative Law Judge determined that the licensee should conduct a population-based study. The Judge's principal concern with IFIM is that the relationship between habitat and population has not been "persuasively demonstrated." The biological study approved by the Judge, however, requires the licensees to study only two flow regimes and requires each regime to be studied for only one life cycle.

This initial decision has been appealed to the full Commission by the intervenors who believe that the record fully supports IFIM and that the population-based study approved by the Judge will fail to provide any basis for determining minimum flows. It should be noted that the FERC Staff fully supports the use of IFIM in this case.

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* Edward F. Lawson is an attorney in the firm of Koff and Lawson, Boston, Massachusetts.

Recent Publication*

Striped Bass Paper Available

A paper prepared by Johnie H. Crance*, entitled "Habitat Suitability Index Models and Instream Flow Suitability Curves: Inland Stocks of Striped Bass," (FWS/OBS-82/10.85) has been submitted for publication to the Government Printing Office by the U.S. Fish and Wildlife Service, Western Energy and Land Use Team. The paper was developed by synthesizing facts, concepts and opinions obtained from published and unpublished reports, a Delphi panel of 18 striped bass experts/authorities, and the Striped Bass Committee, Southern Division, American Fisheries Society.

The Suitability Index (SI) curves and the habitat use/preference information in the paper are potentially useful for stream analyses using the Instream Flow Incremental Methodology (IFIM) and for water resource planning and management where striped bass are a concern.

The paper contains an evaluation form with a request for reviewers and users of the curves and models to provide feedback useful for updating and reprinting the SI curves and habitat suitability information.

Address requests to: U.S. Fish and Wildlife Service Instream Flow Group, 2627 Redwing Road, Fort Collins, CO 80526-2899 (Phone 303-226-9318 or FTS 323-5318).

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*Johnie Crance is a Fishery Biologist, WELUT, IFG.

Legal Institutional Analysis Model Used in Alaska

by
Leah J. Wilds
Management Analyst, Instream Flow Group

B.L. Lamb and Leah J. Wilds of the Instream Flow Group's Institutional staff made a recent trip to Alaska as consultants to Habitat Resources personnel at the Fish and Wildlife Service, Region 7 in Anchorage, Alaska. Region 7 was involved in preparing for participation in a State resources management conflict. For the first time, the Legal-Institutional Analysis Model (LIAM) was used to scope a specific resource problem; based on the results of the model, Region 7 was able to develop an understanding of the issues and organizations involved, as well as the political and legal aspects of the conflict. Based on this experimental application of the model, it appears that LIAM does allow the user to better prepare for participation in resource-related actions and to develop initial and alternative negotiation strategies. This is exciting news for those involved in its development, as well as for potential users. Region 7 personnel indicate that the model, with further refinement, will likely become an important and relevant management tool in future problem-scoping situations. Training in the use of the model, as well as negotiation techniques is provided in IFG 310, "Application of Technical Information in Decision-Making."

IFG Training

IFG 210 Using the Computer Based Physical Habitat Simulation System (PHABSIM)
March 25-28, 1985 U. of Washington, Seattle
Instructor: Robert T. Milhous
April 15-19, 1985 Fort Collins, Colorado
Instructor: Ken D. Bovee
Robert T. Milhous

This 40 hour course provides "hands-on" training in the use of the library of computer programs in the PHABSIM system. Activities are divided between morning lecture sessions and supervised afternoon exercises on the computer. This course is intended for: 1) persons responsible for processing field data through PHABSIM system models; 2) project leaders and others primarily responsible for the field measurements required of a complete stream habitat analysis; and 3) those responsible for quality control, or those directly or indirectly responsible for analyzing, interpreting, and defending the results of a study. Introductory concepts and use of IFIM are not covered. Materials provided include the user's documentation to the PHABSIM system, a detailed problem example, and a primer on computer usage. Each class is limited to 20 students who are divided into groups to give everyone working experience with the computer. No prior computer experience is necessary. Prerequisite: IFG 200. Tuition: Public $450; Private $550.

IFG 315 Advanced Analytical Techniques in IFIM
Temperature Modeling
January 21-25, 1985 Fort Collins, Colorado
Contact: John Bartholow

This 36 hour course provides "hands-on" training in the use of the instream temperature model. The Instream Flow and Aquatic Systems Group in cooperation with the Soil Conservation Service, has developed a model to predict instream water temperatures for historical or synthetic hydrological, meterological, and physical stream geometry conditions. The model is applicable to any size watershed or river basin. It incorporates many features including: heat transport - to predict the average daily water temperature as a function of stream distance; heat flux - to predict the energy balance between the water and its surrounding environment; solar - to predict solar radiation penetrating the water; shade - to predict the solar radiation-weighted shading due. to both topographic and riparian vegetation; meteorology corrections - to predict the changes in air temperature, relative humidity, and atmospheric pressure as a function of elevation; regression aids - to smooth and/or fill missing water temperature data at headwater and internal validation/calibration locations. The model can and has been used satisfactorily to evaluate the impact on instream water temperatures for the following applications: various reservoir releases; riparian vegetation; stream withdrawls and returns. This course is recommended for biologists and engineers responsible for the analysis of water systems and evaluation of water management schemes. Class size: 20 maximum. Prerequisite: IFG 215 or permission. Tuition: Public $450; Private $550.

IFG 310 Application of Technical Information in Decision Making
Jan. 28-Feb. 1, 1985 Fort Collins, Colorado
Instructor: Berton L. Lamb
Leah J. Wilds

This 32 hour course is designed for those who are proficient in IFIM, or in the use of HEP in instream flow analysis and negotiation. Emphasis is given to the use of the IFG Legal/Institutional Analysis Model (LIAM) in decision making and negotiations. The purpose of this course is to train natural resources professionals in the skills necessary to conduct effective policy analysis, present data to decision makers, and negotiate impacts of water development projects. Offered in alternate years. Registration limited to 35. No prerequisite. Tuition: Public $300; Private $400.

IFG 321 Seminar on Hydraulics in IFIM
April 11-12, 1985 Fort Collins, Colorado
Instructor: Robert T. Milhous

This 16 hour seminar provides advanced discussion and training in the use of hydraulics in the Physical Habitat Simulation System (PHABSIM) element of the Instream Flow Incremental Methodology (IFIM). This seminar is being offered for experienced PHABSIM users who want to discuss and gain experience in using the many options which have been described in the Technical Notes available from the Instream Flow Group. The time allotment will provide 12 hours of discussion and 4 hours of "hands on" opportunities. The April 11-12, 1985 seminar will cover the following topics: the use of IFG4 with one data set; the selection of the hydraulic simulation techniques most appropriate for various PHABSIM applications; use of IFG4 and WSP together; and the development of stage-discharge relationships. Class size: 25 maximum. Prerequisite: IFG 210 or IFG 215. Cost: $100.00.

Course Graduate Comments

Comments from IFG 300, the Water Law Short Course, November 14-16, Fort Collins, Colorado:

> "The speakers were well informed, articulate and entertaining. The background and general discussions were quite interesting and informative. Almost as important was spending three days with a variety of people from different states who were knowledgeable about and interested in the allocation and regulation of water."

> "Good overview of the nature and complexity of water law. It heightened my sensitivity to issues and concerns that my agency will need to address in the near future."

Training Calendar

IFG 305	January	14-18, 1985	in	Leetown, WV
IFG 315	January	21-25, 1985	in	Fort Collins, CO
IFG 320	January	22, 1985	in	Portland, OR
IFG 310	January	28-2/1, 1985	in	Fort Collins, CO
IFG 200	February	11-15, 1985	in	Seattle, WA
IFG 210	March	25-29, 1985	in	Seattle, WA
IFG 321	April	11-12, 1985	in	Fort Collins, CO
IFG 210	April	15-19, 1985	in	Fort Collins, CO
IFG 215	May	13-17, 1985	in	Fort Collins, CO
IFG 200	July	15-19, 1985	in	Fort Collins, CO
IFG 205	July	22-26, 1985	in	Pingree Park, CO
IFG 205	September	16-20, 1985	in	Leetown, WV
IFG 200	October	28-11/1, 1985	in	Leetown, WV
IFG 315	November	4-8, 1985	in	Fort Collins, CO
IFG 300	November	13-15, 1985	in	Fort Collins, CO
IFG 215	December	2-6, 1985	in	Leetown, WV
IFG 210	December	9-13, 1985	in	Fort Collins, CO
IFG 210	February	17-21, 1986	in	Fort Collins, CO
IFG 215	May	12-16, 1986	in	Fort Collins, CO
IFG 200	July	14-18, 1986	in	Fort Collins, CO
IFG 205	July	21-25, 1986	in	Pingree Park, CO
IFG 315	February	3-7, 1986	in	Fort Collins, CO
IFG 205	September	15-19, 1986	in	Leetown, WV

To register for courses contact:

Helen White/Caroline Frye
Office of Conference Services
Rockwell Hall
Colorado State University
Fort Collins, CO 80523
(303) 491-6222

Through these quarterly training announcements we attempt to provide three kinds of information: highlights of upcoming courses, a 2 year training calendar, notes and articles by course graduates on their experiences using IFIM, and suggestions or examples of specific component parts such as PHABSIM and LIAM.

This issue highlights negotiations and application of the IFIM. To submit an article, please contact: Kate Brandes, Editor, Instream Flow Group, 2627 Redwing Road, Fort Collins, CO 80526-2899.

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A Discussion of the Critique of the IFIM

(A Discussion of the Mather, et. al. 1985 Note
in the Canadian Journal of
Fisheries and Aquatic Sciences)

by

The Instream Flow and Aquatic Systems Group
U.S. Fish and Wildlife Service
Western Energy and Land Use Team
Fort Collins, Colorado
80526

CONTENTS

Discussion of the Mathur, et. al. (1985) Note on IFIM

INTRODUCTION

A recent note in the Canadian Journal of Fisheries and Aquatic Sciences by Mathur, et. al. (1985) proports to evaluate and criticize the Instream Flow Incremental Methodology (IFIM) based on assumptions attributed to the methodology. Using previously published data from Orth and Maughan's (1982) evaluation of the methodology, Mathur, et. al. (1985) describe how numerous assumptions of the IFIM were false. However, many of the criticisms misstate the assumptions of the methodology and narrowly focus upon but one of the components of IFIM, namely the physical habitat simulation system (PHABSIM). Consequently, the results of their reanalysis of Orth and Maughan's data are misleading because Mathur, et. al. (1985) failed to test the assumptions of the IFIM. The purpose of this memo is to describe the context in which these criticisms were made, to clarify the actual assumptions of the methodology and document support for these assumptions.

BACKGROUND

The authors of this critique are all, one way or another, working under contract for the Philadelphia Electric Company, currently involved in a relicensing of the Conowingo Dam on the Susquehanna River before the Federal Energy Regulatory Commission. Several meetings and hearings were held before an administrative law judge at FERC, during the summer and autumn of 1983 regarding the type of instream flow study to be used to determine release schedules below the dam. The consultants for the applicants proposed an empirical study, setting an "experimental" minimum release below the dam for a period of four years and comparing the response of the fish community with baseline population data. The intervenors in the case (FWS, Pennsylvania Fish Commission, Maryland Department of Natural Resources, and the Susquehanna River Basin Commission) objected to the applicant's study plan on several counts and proposed the use of IFIM as an alternative to the empirical study. Among the objections to the empirical study was a poorly defined and potentially biased baseline (data taken from a fish trap that was accessible to the fish only during low release periods), a severely constrained experimental design (only one or two low flow releases were proposed; the effects of high flows and hydropeaking operations were ignored), an insufficient monitoring period (intervenors felt that the time period should include at least one, and preferably two complete life cycles), and the problems of obtaining an accurate population estimate in a river the size of the Susquehanna (the possibility that the independent variable of the experiment is unmeasurable). The applicant's stated objection to the use of IFIM was that the decision variable for most applications is usable habitat, not numbers of fish. Their primary contention was that habitat is a poor surrogate for population size and biomass.

The beginning of this controversy over IFIM actually goes back to about 1978, when the Susquehanna River Basin Commission attempted to conduct an instream flow study on the Susquehanna below Conowingo. At that time, the IFIM consisted exclusively of the Physical Habitat Simulation System, which was in an early developmental stage. The need to analyze such things as water supply, water quality and temperature, and food resources were recognized, but no described mechanisms were in place to do so. Additionally, the IFG had relatively little experience in working on very large rivers, such as the Susquehanna, so the SRBC received minimal technical support. To confound these problems, the species criteria database for important fishes in the Susquehanna River was essentially nonexistant. Consequently, the first application of the IFIM on the Susquehanna was hardly a rousing success. By today's standards of application, one would have to describe the study as poorly planned and executed, although at the time, it was probably the best anyone could have done. The important historical aspect is that the first exposure the applicant and consultants had with the PHABSIM was discouraging. In fact, it may have appeared that the IFIM was only a weapon used against them to justify an increased flow below Conowingo (unfortunately a perception with considerable supporting evidence). Therefore, it seems that in its "maiden voyage" on the Susquehanna, the IFIM was viewed by both sides, not as a tool, but as a weapon. Our perception is that, to a large degree, this attitude (by both sides) persists today. The modern version of IFIM has never really been given a rational and objective chance, nor has its potential as a problem solving tool been fully explored. (Part of the controversy revolves around the "political" question of whether a problem even exists in the Susquehanna fishery).

During the initial hearings on Conowingo, two professional critiques of the IFIM, similar to the one now appearing in the Canadian Journal, were prepared. The first appeared in the Water Resources Bulletin (Mathur, et. al. 1983), criticizing a study on red shiners, published by Orth and Maughan (1981). In response, Orth and Maughan (1983) objected to the data manipulations performed by Mathur, et. al. (1983) in their reanalysis of the original data. Orth and Maughan state that "Mathur, et. al., apparently misunderstood the assumption (of the relationship between habitat and standing crop), since they tested the relation between standing crop per unit weighted usable area and weighted usable area." In this comment, Orth and Maughan identified a significant scientific error.

A second critique was submitted for publication to the American Fisheries Society in 1983, this time discussing the Orth and Maughan (1982) study of Glover Creek. This critique was nearly identical to the one now appearing in the Canadian Journal of Fisheries and Aquatic Sciences. The 1983 article was rejected by the AFS, but was apparently repackaged and submitted to the Canadian Journal, where it was published in 1985. Orth and Maughan have submitted a response to the Canadian article, again stating that Mathur, et. al. (1985) have misunderstood the assumptions of the IFIM and have incorrectly manipulated the data to support their arguments.

It should be noted that the Conowingo case is currently under appeal (the administrative law judge initially ruled in favor of the empirical study). These facts should be born in mind by the reader: the context of the Mather et. al critiques is probably more adversarial than academic; the applicant (and its representatives) probably felt that they were being manipulated in their first experience with IFIM; and the focus of the criticism is still at the level of the 1978 technology, regardless of statements to the contrary in the critique. It is also important to recognize that some of the criticisms do not address the technical underpinnings of IFIM at all. Rather the critiques discuss the way that IFIM studies are planned and executed, and the way that "validation studies" have been conducted. Again, the critics' opinions of the methodology may have been biased by their initial experience in 1978, and their criticisms of Orth and Maughan's work reinforce their attention to criticisms of 1978 technology. Although Mathur, et. al. (1985) cite recent materials published by IFG, their arguments do not reflect a knowledge of approaches described in these documents. In addition, no evidence suggests that they have conducted an instream flow study using IFIM, nor have they conducted any original research on this subject.

A DISCUSSION OF THE CRITIQUE

Misconception Number One - IFIM is Synonymous with the Physical Habitat Simulation System

In their opening comments, Mathur, et. al. (1985) describe the IFIM as a standard analytical technique for recommending flows for a stream but proceed to describe and discuss only the PHABSIM components. The methodology actually encompasses much more than this. Although the methodoloy is often used for recommending flows in streams, its broader application is in evaluating various water and habitat management alternatives. This distinction might seem trivial, but the purpose of the methodology is to describe existing habitat conditions, compare them with conditions that would exist with various water project design alternatives, and evaluate different management options (Olive and Lamb 1984). It is not to predict fish production, as Mathur, et. al. (1985) strongly imply. The effective habitat component of the methodology has potential for evaluating fish production, but environmentally sound water management decisions can (and, in most cases, must) be made from habitat analysis. If the analyst has done a good job of study design and implementation of alternative analysis, it is a very good decision making tool. The capability to estimate changes in a fish population resulting from changes in a stream's carrying capacity, based on physical/chemical alterations in habitat, is a very desirable goal and should be the focus of future research. A much greater effort by the research community in this area is predicted over the next decade (Trihey and Stalnaker 1985).

The theoretical description of the Physical Habitat Simulation System (PHABSIM) given by Mathur, et. al. (1985) is fairly accurate, but is misleading when discussed as being synonymous with IFIM. The reader of their article is given the impression that the IFIM consists primarily of PHABSIM. Such a conclusion could not be supported based on IFG publications dated after 1980. Although this microhabitat component model is integral to IFIM, it is only a small piece of the overall analytical framework provided by the methodology.

Misconception Number Two - Interactions Among Habitat Variables are Not Considered in IFIM

The authors cite only one of four possible algorithms for the calculation of weighted usable area in PHABSIM:

$$WUA(i) = C(i) \times A(i)$$

where C(i) is a composite suitability function for each small cell (i), and A(i) is the surface area of each cell, and C(i)= f(v) x f(d) x f(s), f(v)=a suitability weighting factor for the velocity in cell i, f(d)= a suitability weighting factor for the depth in cell i, and f(s)= a suitability weighting factor for the substrate and/or cover in cell i. There are three other options within the PHABSIM system by which C(i) can be calculated. These are discussed in Bovee 1982. The first is to take the geometric mean of the three weighting factors:

$$C(i) = (f(v) \times f(d) \times f(s))^{**}.3333$$

The second approach is to use the smallest value of any of the weighting factors as C(i). The third is an option which allows the user to describe C(i) as an equation, usually in the form of a multivariate exponential polynomial, that encompasses the breadth of a probabalistic analysis of species behavior when such detail is available.

In the early stages of development of PHABSIM, the suitability index curves used to compute C(i) were incorrectly equated with probability functions. Mathur, et. al. (1985) are correct in their assertion that these are really weighting factors, and not univariate probabilities. They further state that calculation of a joint probability function by multiplication of the univariate preference factors is valid only when probabilities are statistically independent. The concept of independence among variables is one that has created a great deal of confusion for many people. The problem is that there are often correlations between two or more variables in the streams under study. Furthermore, fish may select microhabitat sites on the basis of interactive behavior; that is, the range of one variable used by the fish is conditioned by another variable. Where many people are confused is by misinterpreting a correlation between physical variables, particularly depth and velocity, and attributing the physical interaction as being important to fish behavior. An example of biologically induced interactive behavior is exemplified by species which use shallow water in the presence of overhead cover and deep water in the absence of overhead cover. Another example is the use of a multitude of substrate types in slow water, but only large substrate types in fast water. Such types of interactive behavior are routinely incorporated in PHABSIM analyses.

Correlations between depth and velocity are sometimes apparent in the data bases used to construct habitat suitability criteria. However, these correlations have a physical basis and not a biological one; they appear in the data base only as an artifact of the stream in which the fish were observed. The cross-product between depth and velocity is meaningless to the fish. Such correlations occur as a result of developing the criteria data base in a hydraulically simple channel, a practise that is strongly discouraged, but happens nonetheless. Such correlations are not apparent when data bases are developed in streams with a high diversity of habitats. In fact, when developing multivariate preference functions (i.e., correcting habitat utilization probability density functions by dividing by habitat availability probability density functions), we have repeatedly found that the interactive terms between depth and velocity cancel each other out (Voos 1980). The evidence is fairly strong that fish select their preferred depths and velocities independently and not on the basis of the interactions between the two variables. Fish select velocities on the basis of how fast the water is moving, not by how deep it is. Strong cover or substrate preferences for certain life stages of fishes can alter selection of velocities and depths and are handled by "conditional" criteria. The criticism does illustrate the need for more attention to be paid to the development of comprehensive study plans for criteria research studies.

Misconception Number Three - Suitability Index Criteria are Meant to Reflect Actual Probabilities of Fish Occurrence

Mathur, et. al. (1985) also criticize PHABSIM for the use of normalized suitability indexes, rather than actual probabilities. The main reason that the functions are normalized is because the function of PHABSIM is to describe suitability by calculating the amount (i.e., surface area throughout a stream segment) of usable microhabitat at a given discharge. If an area of stream is completely satisfactory to a species, as microhabitat, then the entire area should be counted. This is possible only if the maximum index value describing the suitability is unity. An area of suitable habitat does not become less suitable simply because there is only a 30% chance of finding a fish under optimal conditions.

One of the comments in Mathur, et. al. (1985) regarding the use of normalized vs. probabilistic suitability indexes was that the calculated weighted usable area for one species or life stage is not equivalent to that for another, in terms of the number of fish they will respectively support. This is true, but in 99% of the applications of the IFIM to water management decisionmaking, this argument is irrelevant. Alternatives are evaluated in IFIM by developing a baseline of total usable habitat for each life stage. The habitat baseline is determined from several habitat vs. discharge functions (depending on monthly or seasonal shifts due to different microhabitat utilization, water quality, temperature, or other factors) and a baseline streamflow time series. Alternative flow regimes are then evaluated by changing the discharge time series, and developing a new habitat time series, using the same habitat-discharge functions. The goal of this analytical procedure is to minimize habitat losses for those life stages and species known or considered to be most important. Thus, a water management scenario that increased the habitat for adult smallmouth bass by 10% and for adult carp by 30% would be better than one that reduced bass habitat by 25% and increased carp habitat by 50%, regardless of how many carp or bass could be produced under the second alternative. The point is that in applications of the IFIM, the habitat time series for a life stage or species is compared with other time series, simulated for various water management scenarios, for the same life stages and species. The ability to conduct experiments by gaming with the system allows the investigator to determine how much manipulation of the flow regime can be allowed without reducing the amount of suitable habitat below baseline conditions, or to quantify such reductions under different operating regimes. The decision to favor one species over another is strictly a management decision.

However, it sometimes becomes necessary to evaluate the habitat of one life stage vs. another for the same species. This problem has not been as common as we thought it would be; the same flow events often have the same effect on all the life stages. Techniques are presented in Bovee (1982) that assist the user in determining which life stages and habitat events are most important to protect, if differential habitat changes are predicted under a proposed management scheme.

<u>Misconception Number Four - Fish Population and Habitat Should be Correlated at All Times</u>

The habitat time series is central to both the application and validation of the IFIM. Because of the dynamic hydrologic character of rivers, the habitat for fish and macroinvertebrates is likewise very dynamic. This is often a difficult concept for biologists who have worked in ponds and lakes their entire lives to comprehend. In a river, the amount of habitat expands and contracts monthly, daily, and sometimes hourly. It is during those periods when usable habitat for a species reach their minima that the population is at its greatest density and individuals experience the greatest amount of stress. It seems odd that biologists who acknowledge that growth, recruitment, and mortality are density dependent factors, might have difficulty in accepting the fact that usable living space controls density in streams. This constraint on usable space is not continuous. In some drainages, it occurs during summer low flow months when limited physical space and deteriorating water quality conditions occur. In other streams, it occurs during high flow events associated with floods or hydropeaking operations. At other times during the year, there may be excess habitat available to the fish population if one or more life stages were limited by an episode of low habitat.

The concept of limiting habitat episodes, therefore, should be the focus of "validation studies" concerning IFIM. The assumption inherent with use of the IFIM is that the biomass of a species, (or a guild of species having similar habitat requirements), in the absence of fishing mortality or stocking, can be positively correlated to the limiting habitat events for one or more of the life stages. Mathur, et. al. (1985) state,"The basic foundation of IFIM...assumes a positive linear relationship between WUA and standing stock of fish..." Although these two statements appear to be quite similar, there are major fundamental differences that we believe have led to the authors' conclusions that WUA is unrelated with biomass. Adherence to the assumption as stated by Mathur, et. al. (1985) would require that the amount of suitable microhabitat be instantaneously correlated to biomass at all times in order for the IFIM to be "valid." This would cause no problem to researchers if fish populations could rebound as rapidly as habitat can increase following a limiting habitat event. Without an examination of historical habitat events expressed as a habitat time series, studies of single point measurements of habitat and biomass, made simultaneously, can be very misleading.

The confounding problem is that the limiting habitat episode can be of relatively short duration and may have occurred several months or years before the biomass estimate is made. The presently measured habitat, therefore, would have little relationship to present biomass. Our experience suggests that the likelihood of any population being measured at the time that the habitat is most limiting to that population, is fairly remote. Simulation or back calculation of habitat events (historical habitat time series), year class strength, and growth are all important precursors to any validation study.

The study conducted by Orth and Maughan (1982) is used by Mathur, et. al. (1985) to demonstrate the lack of correlation between WUA and biomass. The original study was conducted in a small Oklahoma stream during 1978 and 1979 and examined the microhabitat component of IFIM only. Habitat areas for four species of fish were measured quarterly in two pools and two riffles, over the two years of the study. Population and biomass estimates for each species were made at approximately the same time of the year as the habitat measurements at each of the study sites. Habitat events during intervening times and for previous years were not simulated. Nonetheless, Orth and Maughan (1982) found significant correlations between WUA and biomass during the summer low flow period for orangethroat darter, the freckled madtom, and the central stoneroller. They concluded that the evidence supported the hypothesis that a limiting habitat episode limited the abundance of these species. Mathur, et. al. (1985) found no such relationship because they pooled all habitat events for all time periods. Since such an analysis does not test the hypothesis of a limiting habitat event, and includes periods of excess habitat, no correlation with pooled data can be expected.

One of the more interesting aspects of the Oklahoma study, however, is the relationship between the quarterly "snapshots" of the habitat (small samples within an historic habitat time series) and the sampled smallmouth bass population. Orth and Maughan (1982) plotted quarterly microhabitat per unit surface area (essentially an average composite suitability index) and biomass per hectare estimates for adult smallmouth bass, and found no significant correlation for any one time period. From examination of the summer data for smallmouth bass (Figure 1), it is obvious that the four data points for 1978 lie almost on a straight line ($r=0.997$, $p<0.005$). No such relationship is apparent for the same time period during 1979. The maximum biomass measured during the summer of 1979 is less than 25% of the maximum biomass measured during the summer of 1978. This phenomenon suggests that some significant habitat event, possibly unmeasured, occurred between the 1978 and 1979 measurements.

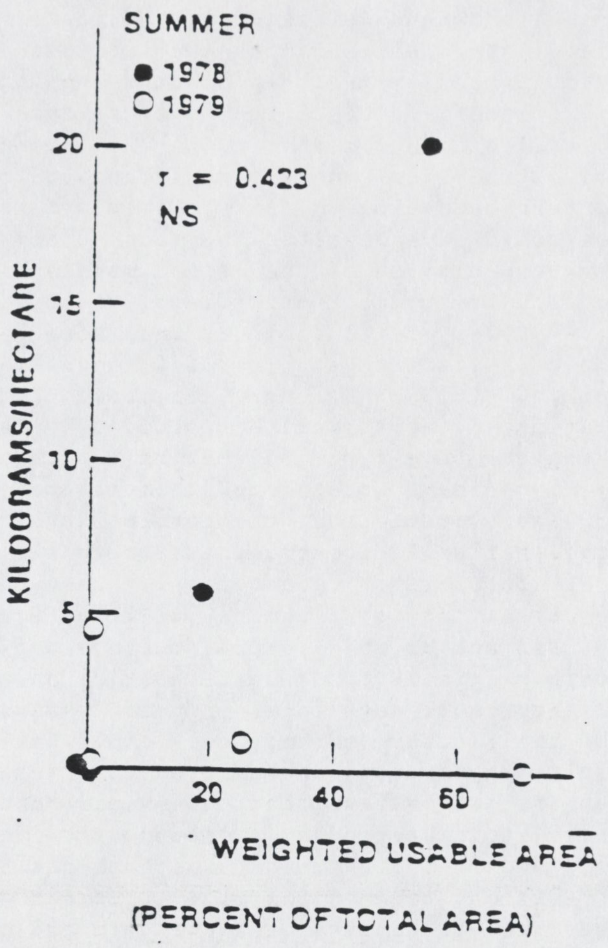

Figure 1. Relationship between WUA (as a percent of surface area) and adult smallmouth bass biomass in Glover Creek, Oklahoma, during the summers of 1978 and 1979. From Orth and Maughan (1982).

Examination of U.S. Geological Survey streamflow records revealed that the streamflow in Glover Creek dropped from 35 cubic feet per second at the beginning of July to 0 cfs by the end of July in 1978. Streamflow did not resume for a period of four months. It is impossible to reconstruct exactly what happened during the one year period between July 1978 and July, 1979. However, gaging records suggest that the streamflow was dropping significantly when Orth and Maughan collected their summer 1978 data. It is possible that some smallmouth bass moved out of the less suitable areas during this time period and found refuge in the two pools used as study sites. This hypothesis, unfortunately, can not be tested, because of the small size of the study sites measured; neither can the habitat dynamics of the stream itself be reconstructed from Orth and Maughan's data. If the pools were at carrying capacity, and the habitat area reduced to an absolute minimum, a high correlation between simultaneously measured habitat and biomass would be expected, and may explain the 1978 results. However, once in the pools, the fish were trapped for at least four months.

During this period, they could have been subjected to higher than normal predation, they could have been exposed to increased, or more effective fishing pressure, they could have depleted the food supply in the pools, or the food supply in unmeasured parts of the stream may have been depleted due to habitat losses for food organisms. Regardless, it is apparent that by the winter and spring of 1979, very few smallmouth bass were left in the stream reach. A nearly identical situation is described for a small stream in Illinois (Larimore, et. al. 1959). Larimore, et. al. noted that the smallmouth bass population in the stream they studied, remained depressed for an entire year following the same type of limiting habitat phenomenon.

Mathur, et. al. (1985) have interpreted the smallmouth bass results from Orth and Maughan (1982) as evidence that the concepts underlying the IFIM are invalid for warmwater fisheries. However, they lumped all the data from all seasons into one regression; this is scientifically unsound and does not test the methodology. The actual habitat dynamics in Glover Creek are not well enough defined by the Orth and Maughan (1982) study to definitively test the limiting habitat concept. This is unfortunate, but certainly not the fault of Orth and Maughan. If they could have foreseen that a major drought would occur in the middle of their study, perhaps they could have designed their study differently. In the context of the habitat time series and the concept of limiting habitat events, we believe that the events occurring in Glover Creek during the 1978-1979 time period do not support the conclusions of Mathur, et. al. (1985), and in fact, support the concepts of the IFIM.

Time lagged correlations between a limiting habitat event and the observed year class strength or biomass of adult fish are probably quite common. Such analyses are more appropriate as the basis for "validation studies" than are instantaneous habitat and biomass measurements made at the time the adult life stage is limited by some predetermined, regularly occurring habitat minima. We suspect that it may be quite common to find that the adult biomass measured in one year is correlated with the amount of spawning habitat or fry habitat that was available several months or years earlier. An excellent example of such time lagged correlations have been found by Nehring and Anderson (1985). Similar phenomena have been documented by Loar, et. al. (1985) and by Gowen (1984).

COMMENTARY ON THE CRITIQUE

As mentioned in the "Background" section, it is important to consider the context in which the Mathur et. al criticisms of the IFIM were made. The IFG welcomes constructive criticism because it invariable leads to advancements in the state of the art and improvements to the tools we develop. However, it has been our experience that constructive criticism can only be made after the researcher has developed his or her own background in using IFIM. The most vociferous, and usually nonconstructive criticisms nearly always come from those who have little or no working knowledge of the methodology. In short, the more people work with IFIM, the more readily they accept it as a decisionmaking tool. We have also noticed that first impressions are extremely important. If a user or project proponent has a satisfactory experience after their first application of the methodology, they are usually quite willing to use it again. However, if their first encounter with IFIM has the appearance of being manipulative, if the study design is faulty, or if there are other undesireable aspects to study implementation, they will not likely accept the methodology from that point on.

It should also be recognized that as long as agency biologists are effective in quantifying project impacts and in influencing water management decisions, the methods they use will always be criticized. One of the primary reasons that the IFIM has drawn so much attention is that it has helped agency biologists to increase their effectiveness in the water management arena. Criticisms of the IFIM can be divided into several categories:

1. Criticisms of the theory and assumptions of the methodology;
2. Criticisms of the output variables and interpretations, thereof;
3. Criticisms of study design and implementation; and,
4. Criticisms of supporting research and the state of the art.

Many of the critiques of the IFIM do not make this distinction and are quite indiscriminate of the basis of their evaluations. For example, Li (1985) objects to the flow recommendations obtained using IFIM because they "often exceed the natural water supply of the stream." This is impossible if the IFIM is correctly used and interpreted. Li's complaint is not germane to the IFIM, but to the way that some users choose to interpret one component. Shirvell (1985; in preparation) suggests that cover is not routinely used in PHABSIM analyses. While this might be true, in some cases, the fact is that cover analysis has been incorporated in PHABSIM software, and taught in IFG courses for at least five years. If users choose not to use cover in their analysis, the IFIM can hardly be held as the reason.

Most of the criticisms in the Mathur, et. al. (1985) article are either trivial or baseless. Their objection to the use of univariate weighting factors instead of multivariate probability functions is irrelevant. First, we have developed dozens of multivariate probability density functions at IFG over the past few years. In virtually every analysis, the cross-product term between depth and velocity (as found in the habitat utilization function) is cancelled out when the function is corrected for habitat availability. In short, the evidence is fairly conclusive that physical interactions between depth and velocity are meaningless to the fish. Orth and Maughan (1982) did not make this correction, so arguments about IFIM validity based on their study are groundless. The capability to evaluate biologically important interactions, such as the selection of certain depths as a function of cover type, or velocities based on substrate size, has been available in the PHABSIM component of IFIM since 1981. Furthermore, should it be determined from empirical data that a biologically important interaction between hydraulic variables does exist, an option within the PHABSIM program allows the use of a multivariate suitability function instead of univariate curves. Most people do not use this option because it is more difficult than using curves, and research to date suggests that it adds little to the precision or quality of the habitat time series output.

Mathur, et. al. (1985) criticize the IFIM on the basis that the output variable is usable habitat, and not pounds of fish. They use the results of the Orth and Maughan (1982) study to demonstrate that there is no relationship between WUA and biomass, even though the original authors concluded otherwise. The fundamental problem with the Mathur, et. al. (1985) reanalysis of the PHABSIM data is that they did not test the limiting habitat concept of the IFIM. It is precisely on these grounds that Orth and Maughan have objected to the misuse of their data by Mathur, et. al. (responding to the three aforementioned critiques). The reanalysis of the Glover Creek data as presented by Mathur, et. al. (1985) is without merit, and emphasizes their fundamental lack of understanding regarding the IFIM. We believe that the Glover Creek data, and the evidence of a major habitat limitation during that study, support the concepts of the IFIM rather than refute them. Furthermore, it seems that Mathur, et. al. (1983,1985) are not content to interpret data as they are collected by the authors. Their persistence in manipulating data to support their own arguments is becoming too commonplace to be considered accidental. There is certainly nothing wrong with bringing in new evidence, or in reviewing or interpreting data by different approaches, but when original data are reformatted and manipulated without stating the reason for the change or the new hypothesis being tested, the results and conclusions can be extremely misleading.

General Commentary on IFIM as a Decision Making Tool for FWS Personnel

There are several aspects regarding the use of habitat as an output variable that should be explored. First, the contention by Mathur, et. al. (1985), that habitat is unrelated to biomass has been found to be false so often that the statement hardly bears repeating. However, it must also be stated that habitat is not limiting at all times in streams. The most basic premise in applying the IFIM is to first search for limiting habitat events through historical baseline habitat time series simulations, and secondly to prevent habitat reductions during the most critical time periods for the life stage or organism most severely limiting the population. The more an investigator knows about the population dynamics and limiting factors of a population, the better the critical habitat limitation can be defined. The less one knows about the population, the more conservative he or she must be in protecting the habitat for all life stages or organisms in a stream. However, the most conservative estimate of habitat requirements using IFIM still acknowledges periods of unused habitat and water. The maximum instream flow recommendation that would be obtained with IFIM typically leaves 50% or more of the water available for development.

While it has been repeatedly shown that habitat limitations can control population sizes of fish in streams (both cold and warmwater), it is true that biomass predictions cannot be made with IFIM in its current configuration without intensive population sampling and population modeling. We feel that such predictions can be made with the inclusion of IFIM as a driving model for a population model. However, we question both the need and the wisdom in doing so in most applications. The development of a habitat driven population model is not terribly difficult. However, the data required to implement such a model would increase the time and cost (already considered to be high by many users) by at least an order of magnitude over conventional IFIM analyses. The second consideration is whether predictions of biomass are in the best interest of overall environmental protection. We suspect that many developers would like to subject instream flows for fisheries to a cost/benefit analysis with power production or other water development. We occassionally hear statements like, "Well, if you can tell me how many fish you'll lose, we'll fill up a tank truck full of them and hand them out to the fishermen." This attitude is hardly in the spirit of environmental protection and mitigation, but unfortunately, is quite prevalent. It is especially noticeable in relicensing of hydroelectric projects, where a power plant or diversion has been in place, draining the river dry for the past 50 years. Developers are quick to argue that it will cost them millions of dollars to produce thousands of dollars worth of fish. They neglect to mention that they have been making money for 30 to 50 years, using a public resource at no cost to themselves. In short, we feel that habitat analysis provides a better overall level of protection than basing decisions on changes in biomass. Such a change in decision variables would elevate the decision process to a strict economic analysis which would probably be much more prone to manipulation and error than the present system.

Mathur, et. al. (1985) have a valid point in that the IFIM has not been tested as thoroughly in warmwater streams as in coldwater streams. The only other "validation" study we are aware of in warmwater streams, was one conducted by Ecological Analysts under contract to the Fish and Wildlife Service. This study was conducted at 12 sites in two Pennsylvania streams,in a very similar fashion to the Orth and Maughan study. The rock bass was selected as the target species because it is reportedly much less migratory than the smallmouth (Gerking 1954), and was thought to be less subject to fishing pressure. Therefore, it was reasoned that the rock bass population would be a better integrator of environmental conditions measured at a given location. A complete habitat time series could not be developed for all the sites, because of a lack of gaging stations near all the locations. Due to a limited budget, the population could not be monitored over a period of years. Despite these limitations, the correlation between summer low flow WUA and summer adult rock bass population was $r= 0.74$ ($p<0.005$). The correlation between summer low flow WUA and adult biomass was $r=0.86$ ($p<0.005$). It is interesting to note that it is fairly common to obtain a better correlation between habitat and biomass than between habitat and population. Apparently, where there is better and more abundant habitat, there are not only more fish, but they also grow better. Long term "time-lagged" validation studies have only been completed on trout streams to date. Such studies have not been conducted on large, warmwater systems. One factor that might help this situation would be to use IFIM to develop a water management plan, then conduct follow-up studies of the fish population to see if the response to the new flow regime agrees with the conclusion made with IFIM from the habitat simulations. Such follow-up studies, although they might be relatively expensive, would provide the kind of insight needed to determine the "validity" of IFIM. Furthermore, they would provide the information needed to improve IFIM concepts, should the measured results differ significantly from those predicted. Until several such studies are completed, continued criticisms can be expected.

The design and implementation of instream flow studies, especially those using the IFIM, has been a growing concern of the Instream Flow Group for several years. The IFG 200 course was specifically designed to orient new users to the capabilities and components of the IFIM, as well as to describe alternative technologies. Unfortunately, many current users began conducting instream flow studies with IFIM many years ago, or have not participated in the IFIM training. Recently initiated studies, where all parties collectively lay out the study plan, have shown that IFIM users have benefitted greatly from taking IFG 200. It is extremely important to start off on the right foot, especially when dealing with someone to whom instream flow is an alien concept.

For example, suppose an IFIM analysis is started by selecting largemouth bass as the target species, and in the study design it is decided that no consideration is given to food supply or water quality. Because the largemouth is a lacustrine species, it will show a strong preference for zero velocity water. This will tend to skew the WUA versus discharge curve over toward zero flow. What the model is saying is that if largemouth bass prefer pondlike conditions, and at zero flow the river turns into a series of ponds, the best physical conditions will occur at zero flow. If, at that time, the investigator decides that water quality, temperature, or food supply would deteriorate at zero flow, thus adversely affecting the bass, it may be too late to insert these characteristics into the model. It would be too late because data have already been collected and much of the decision time period expended. Furthermore, the project proponent might view such a move as changing the rules of the game in mid-course (a view that we probably agree with). This example illustrates the kind of problem that can arise when studies are initiated without a comprehensive planning process. Unless water quality, temperature, food supply or other macrohabitat factors can be categorically omitted from an IFIM study, they should be incorporated at the outset. It is better to study temperature, and find out that it is not a problem, than to ignore it until the first results come in, and then try to redesign the study. The same concept holds for all IFIM components.

Project developers should also be made aware that there are other methods that can be used to develop instream flow recommendations, and that these are universally more conservative (i.e., recommend more discharge) than the IFIM. The highest flow recommendation one could make using the IFIM would be the median monthly flows. If anyone makes a higher recommendation by taking the peak of the WUA versus discharge function, they are not using IFIM. They might be using some of the component models, but instream flow recommendations that exceed the natural water supply, violate one of the most fundamental principles of the methodology.

If an alternative methodology is chosen instead of the IFIM, it is important that the precepts of those methods be followed as closely as those of the IFIM. For example, some investigators have used a wetted perimeter approach in lieu of the IFIM. However, transects were averaged throughout a study site set up as a representative reach, such as one would establish for a PHABSIM study. This constitutes a misapplication of the method because the wetted perimeter method is based on a critical transect concept. To apply the wetted perimeter method correctly, the user should search out the widest, shallowest cross section, and develop the wetted perimeter versus discharge function at that single critical point. Failure to do so would invalidate the application.

Conclusions

There are several points to consider with respect to critiques of the IFIM, such as those expounded by Mathur, et. al. (1983,1985). First, these criticisms appear to be self-serving, rather than constuctive academic critiques. This perception is substantiated by the authors' continued manipulations of other researchers' data and their tendency to misstate the hypotheses and assumptions of the IFIM. Such misstatements cannot be attributed to ignorance, because the underlying principles and assumptions of IFIM were explained in testimony at an FERC hearing in 1983, in the presence of the authors. The submission date of the critique in the Canadian Journal was in July of 1984, fully eight months after the hearing. Since the principles and assumptions of the methodology were known to the authors well in advance of the submission of their critique, their article which misstates the assumptions of IFIM can only be suspect.

Second, the results of the Orth and Maughan study (1982) and events surrounding that study, plus the rock bass study in Pennsylvania, and numerous studies in cold water streams support the concept of habitat-limited populations. These studies have demonstrated several phenomena:

1. That the IFIM can accurately reproduce the quality and quantity of habitat available over time;

2. That habitat limitations can and do limit fish populations and biomass;

3. That these habitat limitations are of limited duration, and do not affect the same life stages, species, and food organisms the same way;

4. That there is typically an excess supply of habitat during portions of the year in natural streams;

5. That correlations between habitat and biomass must be "time lagged" to the habitat event or events that established year class strength; and,

6. That IFIM is not omniscient. Failures to obtain correlations between habitat and biomass are more often the result of the design and implementation of the "validation study" (and usually the consequence of not identifying the limited life stage or the limiting event) than a fallibility in the logic of the IFIM. If the investigator does not test for a limiting habitat type or event, or tests for the wrong one, failure to find a correlation does not invalidate the IFIM.

Third, project applicants should be made aware that the IFIM is not their only choice in terms of instream flow methodologies. They should also be made aware that all other methods, if correctly applied, have been found to be more conservative in their estimates of instream flow requirements than the IFIM. Furthermore, alternative methods should be required to undergo the same intense scrutiny as the IFIM has undergone over the past eight years. For example, the empirical study proposed for the Susquehanna might, with several important modifications, prove to be a scientifically acceptable study. However, the study could cost the applicant as much as $6 to $12 million, and will test only one or two minimum flow releases. The effects of high flows and hydropeaking operations will not be tested, at all. The IFIM can be used to test more flow regimes in a month than could be tested empirically in a century, and at a fraction of the cost.

Finally, the IFIM is a complex, interconnected system of procedures. This is an inevitable consequence of trying to predict the dynamics of a complicated environment. It is also the result of the IFG responding to suggested improvements and constructive criticisms. Unfortunately, as the methodology is used more and more in warmwater systems, this complexity is expected to increase (for example, a conceptual habitat model incorporating interspecific competition already exists, but has not been developed as an operational model, yet). The IFG is concerned about this increasing complexity, not from a development standpoint, but from an operational one. It requires a committment by the user community to keep abreast of the state of the art. Many of the criticisms levelled at the IFIM should more appropriately be directed at the study plan and implementation of the methodology. Some of the simplistic studies that have been reviewed by the IFG over the past several years have been a constant source of consternation. Even more disconcerting has been the lack of IFG and FWS Ecological Services involvement in the development of these study plans, either through formal training or through direct technical assistance. By the time the IFG has been involved, that critical first step has been taken and whatever perceptions the project proponent has of the IFIM or its implementation have already been made. There are two possible solutions to this problem. First, it is incumbent upon the user to obtain the basic IFIM training (IFG 200-215 and IFG 310) and to update that training periodically, usually through one of the 400 level courses. It follows that the user should then attempt to apply what is taught in those courses. The second solution is simply to call the IFG, before a study plan is initiated. The IFG is available for on-site techical assistance; the only expense to the field office is for travel; or your technical team can be sent to Ft. Collins for consultation. However, the demand for IFG time for technical assistance is great, so potential technical assistance requests should be anticipated by at least six months, if not a year.

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DRAFT

INSTREAM FLOW METHODOLOGIES

RESEARCH PROJECT 2194-2
DRAFT REPORT

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PART I: INTRODUCTION

PURPOSE AND CONTENTS OF THIS REPORT

The purpose of the research contract of which this report is the principal product, was to collect, summarize, and review existing methods useful for predicting the ecological consequences of altering natural stream flows. Many of these methods are known as "Instream Flow Methodologies", and others as habitat suitability or habitat quality indices.

Instream flow is a term applied to the water remaining in a stream or other natural watercourse downstream from a dam or diversion structure. The volume and timing of instream flows are partly and sometimes completely, under the control of the operator of the dam or diversion, but the volume and schedule of releases most convenient and useful to the operator may not be those most beneficial to the stream ecology.

The central premise of many of the methods reviewed here is that stream ecology can proceed in an acceptable manner with less than the full natural flow of the stream, and that there is a way to discover just how much of the natural instream flow should remain and when. None of these methods attempts to strike a balance between the value of instream flow and conflicting water uses (such as the generation of electricity or agricultural irrigation), however, and many of them are explicitly designed to discover the flows which maximize fish habitats.

In the course of reviewing the existing methods, it became apparent that although they represented a broad range of approaches to ecological modelling, there was seldom any mention of why a particular approach as chosen, and more importantly, why other approaches were excluded.

It seemed appropriate, therefore, that this review should include a discussion of the various generic approaches used and how they fit into the range of models possible.

This report consists of two principal parts: Part I begins with an introductory chapter which traces the development of these methods and points out some of the broader categories into which they can be subsetted. The following three chapters describe the various considerations implicit in all of the methods, and are intended to constitute a primer on this type of ecological modeling. Chapter 5 describes the sensitivities of some of the methods to variations in input data and the problems associated with inappropriate model application. Chapter 6 presents data from model validation studies and comments on the status of validation of the various methods. Chapter 7 presents guidelines for choosing methods and describes the ways that conclusions about appropriate flows can be drawn from them, and Chapter 8 presents the conclusions of the authors of this report.

Part II consists of one-page Methodology Summary Forms which encapsulate, in a uniform format, each of the methods reviewed in the report. We hope that the Methodology Summary Forms will be useful and will give a broad sense of the range of methods currently in use.

Finally, there is an annotated bibliography of the papers we have found most useful in preparing this report, all of which are cited in text or in the Methodology Summary forms.

CHAPTER 1: GENERAL OVERVIEW

The specific purpose of this report is to review methodologies that are designed to determine the amount of water that should be left in a stream (the instream flow) when the stream is altered by impoundment or diversion. The principal focus of most such methods has been the minimum acceptable flows although the general topic includes the maximum appropriate flows, and the rate and frequency of changing flows. Furthermore, most such methods have been restricted to the effects on fish, principally salmonids, with an occasional paper devoted to the effects on non-salmonids, stream invertebrates, vegetation, and recreation.

In addition to methods and models specifically designed to recommend an instream flow, there is another class of models which include either flow, or flow-related terms but which are designed to explain or predict habitat quality or fish standing crop. The more we examined these latter models, the less distinction we were able to see between them and the models aimed directly at choosing instream flows, and as a result we have incorporated many of them into this review. The criterion for including them was the presence of flow or flow-related terms such as mean annual discharge, or depth and velocity. Occasionally, for completeness, we included models that were part of a relevent series even though the particular model had no flow-related terms. Some of the U.S. Fish and Wildlife Service Habitat Evaluation Procedures fall into this category. We also included a few models which seemed to demonstrate the lack of importance of flow-related terms by their ability to predict standing crop in the absence of flow terms. The terms "method", "methodology", and "model" are used interchangably in this report, and a complete list of the ones reviewed is presented in Table 1-1.

The instream flow methods, in particular, have received a good deal of attention from reviewers as well as modelers, and this report owes much to four reviews. Stalnaker and Arnette (1976) made a detailed and highly informative review of the methods then available and laid the groundwork for the development of U. S. Fish and Wildlife Service Instream Flow Incremental Methodology (IFIM), subsequently presided over by Stalnaker. Wesche and Rechard (1980) published an excellent compilation and review of 16 of the instream flow methods then in use, and included a discussion of the progress made by the U. S. Fish and Wildlife Service since 1976. Our general approach of summarizing methods was influenced by that report and by a draft manuscript kindly supplied in late 1984 by Kurt Fausch and Mit Parsons (Fausch and Parsons, 1984) reviewing models that predict the standing crop of stream fish from habitat variables. Finally, Loar and Sale (1981) published a well written critical review of instream flow methods that accurately identified many of the shortcomings of the existing approaches.

TABLE 1-1 INSTREAM FLOW AND HABITAT QUALITY METHODS REVIEWED IN THIS REPORT

Author	Name of Method	Species
Annear & Conder 1983	Wetted Perimeter Method	Salmonids
Barber et al. 1980	Diagrammatic Mapping Method	Coho salmon
Binns & Eisermann 1979	Wyoming Habitat Quality Index	Trout
Collings 1974 (all sp.)	Spawning and Rearing Discharge	Anadromous salmon
Dunham & Collotzi 1975	USFS Region 4 Method	Trout
Edwards 1983	USFWS Habitat Suitability Index Model	Smallmouth Buffalo
Edwards 1983	USFWS Habitat Suitability Index Model	Longnose Sucker
Edwards et al 1983	USFWS Habitat Suitability Index Model	Longnose Dace
Edwards et al 1983	USFWS Habitat Suitability Index Model	Slough Darter
Edwards & Twomey 1982	USFWS Habitat Suitability Index Model	Common carp
Geer 1980	Utah Water Records Methodology	Trout
Gilbert 1984	USFWS Habitat Suitability Index Model	Warmouth
Hickman & Raleigh 1982	USFWS Habitat Suitability Index Model	Cutthroat trout
Hoppe 1975	Minimum Stream Flows for Fish	Trout
Inskip 1982	USFWS Habitat Suitability Index Model	Northern Pike
Larsen 1980	USFWS New England Flow Recommendation Policy	All
Layer 1983 (all sp.)	Habitat Suitability in Prarie Streams	6 Warmwater Species
Layher & Maughn unpub.	Habitat Suitability Index Model	Spotted Bass
Li et al. unpub.	Discriminant Habitat Analysis	Cutthroat trout
McMahon 1982	USFWS Habitat Suitability Index Model	Creek chub
McMahon 1983	USFWS Habitat Suitability Index Model	Coho salmon
Milhous et al 1984	USFWS IFG4 Hydraulic Simulation Model	Hydraulics
Milhous et al 1984	USFWS Water Surface Profile Model	Hydraulics
Milhous et al 1984	USFWS HABTAT Model	All Species
Nelson 1984	Montana DFWP Wetted Perimeter Method	All
NGPRP 1974	Northern Great Plains Resource Program Method	All
Nickelson 1976	Habitat Needs for Salmonid Rearing	Coho salmon
Nickelson et al 1979 (all	Stream Flow Requirements for Salmonids	Steelhead, Coho
Orsborn 1981	Spawning Habitat Using Watershed and Channel	Steelhead
Pardue & Cordes 1983	USFWS Habitat Suitability Index Model	Alewife/Blueback Herring
Parsons et al. 1981	Fish Habitat Index Using Geomorphic Parameters	All
Rabern 1984	Habitat Based Georgia Standing Crop Models	9 Species
Raleigh 1982	USFWS Habitat Suitability Index Model	Brook trout
Sams & Pearson 1963	One Flow Method	Anadromous salmonids
Stuber 1982	USFWS Habitat Suitability Index Model	Black Bullhead
Stuber et al. 1982	USFWS Habitat Suitability Index Model	Green Sunfish
Swank & Phillips 1976	USFWS Region 6 Single Transect Method	Salmonids
Swift 1976	Washington Basin Variables Method	Steelhead
Swift 1976	Washington Toe-Width Method	Steelhead
Swift 1979	Washington One-Variable Regression Method	Salmon
Taylor 1982	Riparian Strip Width Model	Vegetation
Tennant 1975	Montana Method	All
Thompson 1974	Oregon Usable Width Method	Salmonids
Trial et al 1983	USFWS Habitat Suitability Index Model	Blacknose Dace
Trial et al 1983	USFWS Habitat Suitability Index Model	Common Shiner
Trial et al 1983	USFWS Habitat Suitability Index Model	Fallfish
Trial et al. unpub	USFWS Habitat Suitability Index Model	Atlantic salmon
Waters 1976	California Instream Flow Method	Salmonids
Weatherred et al. 1981	R2-Cross-81 Sag Tape Method	Hydraulics
Wesche 1980	WRRI Trout Cover Rating Method	Trout
White 1976	Idaho Instream Flow Method	All
White et al. 1976	Midwestern Trout Standing Crop	Trout

In addition to these review papers, many authors sent unpublished manuscripts, some in preparation for publication and others, file reports that are not destined for publication. In the former category were a series of papers comprising the results of a symposium on Validation of U.S. Fish and Wildlife Service Habitat Suitability Index Models, supplied by James Terrell. In the latter category were a series of 14 instream flow evaluation studies funded by the U.S. Fish and Wildlife Service and conducted in many of the western United States (supplied by Clair Stalnaker).

CONVERGING DEVELOPMENT OF INSTREAM FLOW METHODS AND HABITAT QUALITY MODELS.

After reviewing the various instream flow methods and habitat quality models presently available it appears to us that they have been developing along converging paths. The development of both has progressed to the point that it is possible to envision models that can be used equally well both for suggesting appropriate instream flows and for characterizing overall habitat quality at any particular flow. Figure 1-1 summarizes the development of instream flow methods. The diagram is somewhat oversimplified since almost every method differs from every other in some significant way, but it is intended to give a general overview of the approaches use by various authors.

METHODS USING BASIN-WIDE INFORMATION AS INPUT VARIABLES

The types of methods shown in Figure 1-1 are arranged from top to bottom in order of complexity. Methods of the types shown in Figure 1-1a using only or dominantly river basin variables (described in Chapter 2) are not common, but provide an easy way to make recommendations in the absence of any field data. The policy followed in New England by the U.S. Fish and Wildlife Service is of this type (Larsen 1980) and consists of establishing by some means a relationship between basin size (or other basin feature) and recommended flows. Collings (1974) used a similar approach but did it empirically by measuring the amount of discharge that resulted in maximum spawnable area, then used multiple linear regression to correlate this discharge with basin, hydraulic, and structural characteristics. Larsen does not describe the reasoning that led him to the particular functional relationship he recommended and it appears to be arbitrary. Collings used an equally arbitrary but reproducible criterion: The spawning sustaining discharge is is the discharge in which the percentage reduction in spawnable area is just less than the percentage reduction in preferred discharge.

METHODS USING MEAN DISCHARGE INFORMATION AS INPUT VARIABLES

Figure 1-1b illustrates methods using variables related to mean discharge (annual, monthly, daily and monthly minimum discharge, etc.). Tennant (1975) recommended an unspecified (10-60%)

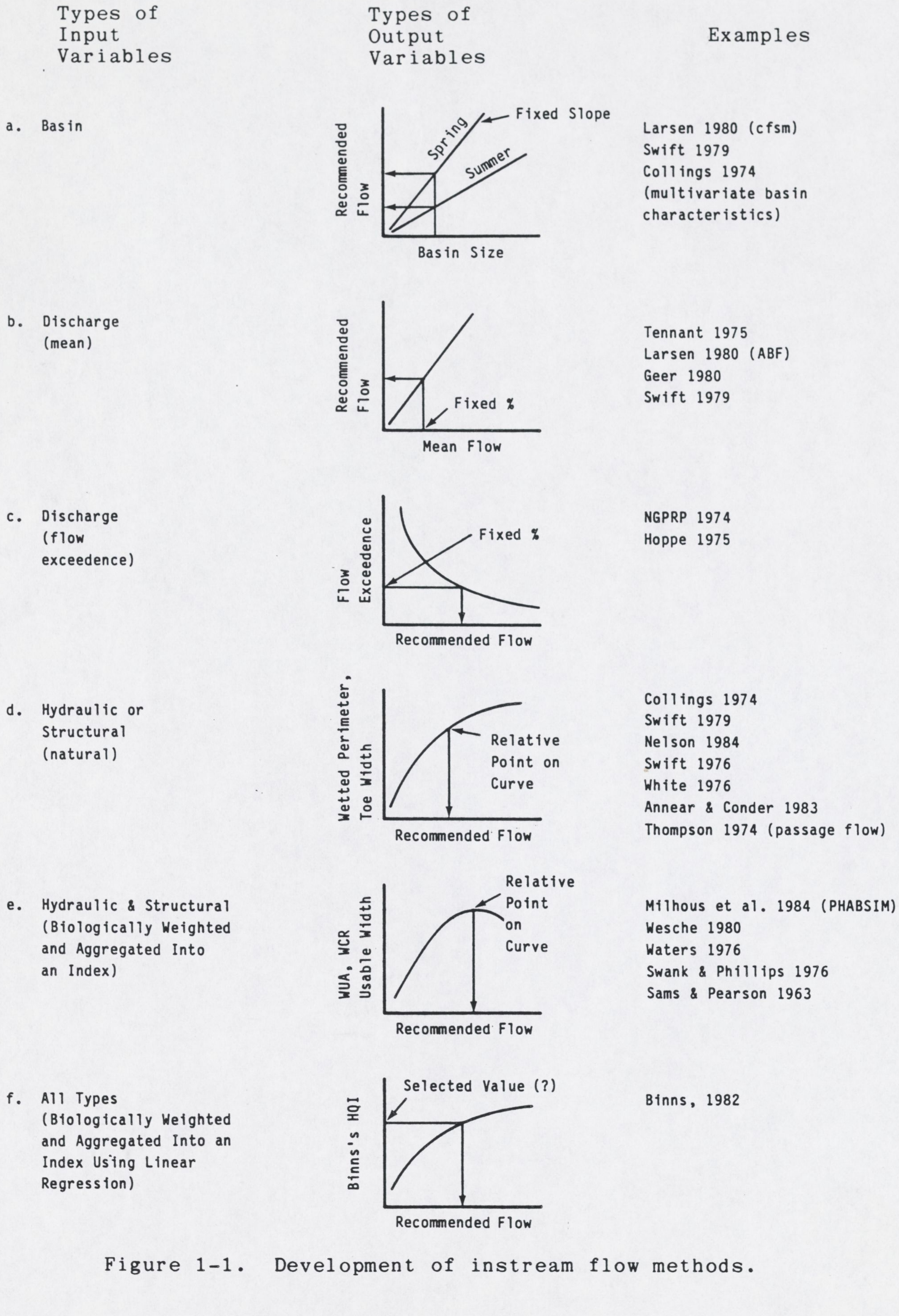

Figure 1-1. Development of instream flow methods.

percentage of the average annual flow, depending on whether the biological outcome desired was minimal or excellent. Geer (1980) suggested using the average 6 month minimum monthly flow for the period of record. Larsen (1980) suggested the 25 year median daily unregulated flow. All of the recommendations are essentially arbitrary, though not necessarily unreasonable. Swift (1979) on the other hand, empirically determined the flow that would maximize spawning, then regressed median September flow and median October flow on it for 28 different streams. This provided him with a formulation with the clear endpoint of maximizing habitat for a particular use.

METHODS USING FLOW-DURATION CURVES AS INPUT VARIABLES

Another discharge approach is illustrated in Figure 1-1c in which a flow-duration curve is used as the basis for recommending flows. The Northern Great Plains Resource Program (NGRPRP 1974) recommended a flow equal to the average daily flow exceeded 90 percent of the time for the period of record excluding months in the lowest or highest 15th percentile of mean monthly flows for the period of record. Hoppe (1975) recommends the flow equal to the average daily flow exceeded 80% of the time as a minimum flow. Both of these recommendations are also arbitrary.

METHODS USING WETTED PERIMETER OR A SIMILAR INPUT VARIABLE

A number of authors, realizing that the discharge variables were not necessarily correlated with any biologically beneficial features in the stream turned instead to simple cross-sectional hydraulic or structural measurements as a way to more closely approximate biological habitat. Figure 1-1d shows the characteristic relationship of two of these variables, wetted perimeter and toe-of-bank width to recommended flow. In the case of wetted perimeter, (the distance from water's edge to water's edge along the bottom) the variable changes with flow and a variety of biological benefits can be ascribed to increasing the amount of wetted area. A curve of the general shape shown can be arrived at from measurements at different flows or from simulation using hydraulic models such as the R2-Cross-81 (Weatherred et al. 1981), the IFG4 (Milhous et al. 1984), the WSP (Milhous et al. 1984) or others. The final step with these methods, is to pick the appropriate point on the curve. White (1976) and Nelson (1984) suggests choosing the "inflection point" on this curve but many of the curves they generated do not have one (see the examples in Nelson 1980). Annear and Conder (1983) suggest that the point on the curve just statistically detectably different from either the mean annual flow or twice the mean annual flow would be appropriate but do not defend the suggestion. Collings (1974) in determining appropriate rearing discharge suggests a point "selected somewhere near" the characteristic change in slope of these curves. Toe width is described by its inventor (Swift 1976) as the horizontal distance "from the point where the streambed and one bank join to the ground surface on the other bank", and is shown in his diagram as the cross sectional width of the water surface at the rearing discharge (defined as the "inflection point" on the wetted perimeter curve). Swift regressed the rearing discharge on toe width to produce a statistic much like wetted perameter, with the same problems associated with picking out the appropriate point.

Swift (1979) subsequently regressed preferred spawning discharge on toe width (as well as on individual basin and discharge characteristics) to provide a whole set of one-variable equations for specifying discharges, which will result in maximum spawning area in the streams he studied.

METHODS USING SEVERAL HYDRAULIC AND STRUCTURAL INPUT VARIABLES

The next change in the development of these graphical techniques, all of which resulted in two-dimensional plots of an index variable against flow, was the inclusion of multiple input variables. In order to retain two-dimensional plots as the data output format these multiple input variables had to be aggregated in some way to form a single output variable that could be plotted as a function of flow. One approach (not illustrated in Figure 1-1) was to form an index by aggregating several basin and discharge variables and scaling its output to correspond to the discharge resulting in maximum spawning area (Orsborn 1981).

Another approach, illustrated in Figure 1-1e, and the one most widely used today, was to select several hydraulic and structural variables (such as depth, velocity, cover), to adjust (transform, weight) their values to correspond to their biological importance, and aggregate them together to form an index.

The simplest of these is Sams and Pearson's (1963) one-flow method in which the average velocity and average depth required over redds were multiplied times width to approximate the discharge needed. Swank and Phillips (1976) used a similar approach but simply identified the distance along a transect meeting both depth and velocity criteria for spawning and called this "Usable Width" which varied with flow. Wesche (1980), also used binary criteria for depth and substrate suitability associated with cover to produce an index, the WCR, which was a function of flow.

Waters (1976) realized that binary depth and velocity criteria like those used by Swank and Phillips were too abrupt and hypothesized that suitability for biological uses was graded. Waters used 4 variables (depth, velocity, substrate size, and cover) and transformed the variables to correspond to the biological importance of the various levels as described in the literature. Raw data were collected from transects for many points and discharges, then biologically weighted, scaled from 0-1, then multiplied together to give, in relative units, the amount of suitable habitat in the stream. Waters then extrapolated these relative units to area producing a plot of area of relative habitat versus discharge.

The U.S. Fish and Wildlife Service Instream Flow Service Group modified Waters' method by including a hydraulic simulation model in the computer program so that the field data would have to be taken at no more than a few discharges. Their set of programs, know as PHABSIM (Milhous, et al. 1984), produces output which includes several hydraulic variables (such as wetted perimeter) as well a variable similar to the one developed by Waters, known as Weighted Usable Area (WUA). The HABTAT submodel of PHABSIM which produces the WUA index is limited to 3 input variables, and current practice is to use weighted depth and velocity plus one other weighted variable such as substrate size, cover, or temperature.

METHODS USING LARGE NUMBERS OF VARIABLES

Instead of developing the Waters type of model in the direction of hydraulic simulation with a limited number of hydraulic input variables, Binns (1979, Binns and Eisermann 1979) applied the concept of biological weighting to a large number (22) of candidate input variables including depth, velocity, substrate and cover. Building on the work of Platts (1976), who observed that control of fish populations was not isolated to any one variable, Binns and Eisermann rated (transformed) all input variables to values of biological importance, then performed one-variable linear regression analyses between each transformed variable (during late summer flow conditions) and standing crop. Those with the strongest correlations were retained and used in a multiple linear regression equation against standing crop. After considerable experimentation and combining of variables (described later in this report) they produced a multivariate regression equation predictive of standing crop (which was also called the Habitat Quality Index). Subsequently Binns (1982) calculated HQI (or standing crop) for the Green River for several different flows, producing a curve of the type shown in Figure 1-1f.

Meanwhile, a number of other investigators were developing methods designed to predict standing crop using both weighted and unweighted habitat variables; Barber et al. (1980) developed a series of regression equations to describe the effects of 10 hydraulic and structural varables on coho salmon standing crop; Nickelson et al. (1979) produced a regression model using depth, cover, and velocity that explained 91% of the cutthroat standing crop, 79% of the steelhead standing crop, and a model that included only pool volume that described 94% of juvenile coho salmon standing crop; Layer (1983) developed a series of regression models using a subset of 15 chemical and structural input variables that explained as much as 99% of fish standing crop; Li et al. (unpublished) used a stepwise discriminant function analysis to classify habitat as to its cutthroat trout populations; Rabern (1984) developed a series of regression models using 21 habitat variables including variables of all classifications that explained as much as 96% of the standing crop depending on species; Taylor developed a riparian strip width model including discharge and basin terms that explained 66% of the existing riparian strip width on undiverted streams, and White et al. (1976) developed a series of regressions using (principally) discharge variables that explained as much as 95% of the variation in standing crop of midwestern trout.

A number of these models have hydraulic and discharge terms, and although they have generally not been shown to be predictive, or their terms causally related to the existing fish populations, the same situation exists for the instream flow models shown in Figure 1-1.

Finally, there is a large group of models (most reviewed here) prepared under the auspices of the U. S. Fish and Wildlife Service Habitat Evaluation Procedures Group that characteristically use more variables than the other models so far described. The ones described in Methodology Summary Forms in Part II include Edwards (1983 a,b), Edwards et al. (1982), Edwards and Twomey (1982), Gilbert (1984), Hickman and Raleigh (1983), Inskip (1982), Layher and Maughn unpulished, McMahon (1982, 1983), Pardue and Cordes (1983), Raleigh (1982), Stuber (1982), Stuber et al. (1982), Trial et al. (1983 a, b, c) and Trial et al. (1982, unpublished). This selection is not complete but gives an idea of the approach being used. The models are constructed of biologically transformed input variables aggregated in a way pleasing to the authors of the models but not according to any formal technique. Most of these models, when tested, turn out not to be correlated with standing crop, but the treatment of individual variables is like that of Binns and Eisermann (1979) and Layer (1983) and it is reasonable to suppose that if empirical aggregation techniques were used the models might be descriptive.

SUMMARY

The development of all of the methods described in this report seems to us to be converging toward multivariate techniques for describing (and eventually predicting) standing crop, riparian strip width, and other measures of biological productivity. Binns (1982) use of his multivariate model as an instream flow model is likely to forshadow considerable development in that direction using all the variables needed to predict standing crop and choosing variables based on their ability to aid in the prediction, rather than choosing them on the basis of convenience or preconceived notions.

The remainder of Part I of this report includes discussions of selected aspects of instream flow model building using examples taken from the primary literature and often including reworking of the data to illustrate our points.

CHAPTER 2. SELECTING VARIABLES

All instream flow and habitat quality methods utilize one or more input (independent) variables to determine the value of one output (response, dependent) variable.

In this chapter, we discuss the types of variables that have been chosen by the various modelers and address the questions of whether the ones chosen are reasonable, necessary, and sufficient to meet the needs of instream flow determination.

RESPONSE VARIABLES:

We have categorized response or output variables in three different ways: First, functionally in terms of the way the output variable can be used; second based on whether the form of the model and hence, the output variable was determined conceptually or empirically; and third, whether the output variable represents a recommendation, or if not, is measurable, or is unmeasurable.

From a functional standpoint, the original type of instream-flow-method response variable is a recommended flow (marked under heading RF in Table 2-1). Thirteen of the methods reviewed are of this type and might be considered the classic instream flow methods. They completely internalize all aspects of the instream flow decision making and, provided with the requisite input data, result in the answer of a single flow, or in some cases different flows depending on the time of year.

Subsequently, techniques were developed to produce response variables ostensibly linked to habitat quality and to display them in a two-dimensional relationship as functions of flow. These are identified under the heading HQF in Table 2-1. In addition, a number of models produce Habitat Quality Indices that are not generally displayed as functions of flow, although could be, and these are identified under HQI in Table 2-1. Of these, many are the Habitat Suitability Index Models of the USFWS Habitat Evaluation Procedures Group, and these are identified under the heading HSI in Table 2-1. Finally, several of the models use standing crop rather than an index as their response variable; and these are identified under the heading SC in Table 2-1.

All methods were also categorized as either being conceptual or empirical in overall approach, and are so identified under the headings C or E in Table 2-1. Conceptual or mechanistic models are defined here as those which are designed, constructed and operated to produce an output variable in the absence of information about the value of the output variable. The models are constructed in a form which is conceptually pleasing to the modeler, usually because it appears to make biological sense. It is then (usually) reduced to a mathematical equation and then sometimes to a computer program if solving the equation would

TABLE 2-1 THE TYPES OF VARIABLES INCLUDED IN THE INSTREAM FLOW/HABITAT QUALITY MODELS REVIEWED

METHOD	GENERAL FEATURES								TYPES OF INPUT VARIABLES (number of each type shown)					
	RF	HQF	HQI	HSI	SC	BW	C	E	Basin	Discharge	Hydraulic	Structural (Cover)	Biological	Other Physical
Annear & Conder 1983	x						x			1				
Barber et al. 1980					x			x		1	3	5		1
Binns & Eisermann 1979		x	x		x	x		x		2	2	2	1	2
Collings 1974 (all sp.)	x							x	3		4	1		
Dunham & Collotzi 1975		x	x				x				2	2	2	
Edwards 1983 (Buffalo)			x	x		x	x			1	1	2	1	6
Edwards 1983 (Sucker)			x	x		x	x				2	2		1
Edwards et al 1983 (Dace)			x	x		x	x				2	3		1
Edwards et al 1983 (Dart)			x	x		x	x				2	2		4
Edwards & Twomey 1982			x	x		x	x				1	2	1	8
Geer 1980	x						x			1				
Gilbert 1984			x	x		x	x		1	4	2	1		4
Hickman & Raleigh 1982			x	x		x	x			1	2	7	2	5
Hoppe 1975	x						x			1				
Inskip 1982			x	x		x	x				2	2	1	4
Larsen 1980	x						x		1	1				
Layer 1983 (all sp.)					x			x	1		1	3		9
Layher & Maughn unpub.			x			x	x		1			1		1
Li et al. unpub.			x					x	1		1	5	1	
McMahon 1982			x	x		x	x				6	5	2	7
McMahon 1983			x	x		x	x					5	2	8
Milhous et al 1984 (IFG4)					H		x				3			
Milhous et al 1984 (WSP)					H		x				4			
Milhous et al 1984 (HAB)		x				x	x				2	1		
Nickelson 1976			x			x	x				1	2		
Nickelson et al 1979 (all)			x			x	x				2	3		
Nelson 1984		x					x				1			
NGPRP 1974	x						x			1				
Orsborn 1981	x						x		2	2	1			
Pardue & Cordes 1983			x	x		x	x					1	1	3
Parsons et al. 1981			x					x	4					
Rabern 1984 (all species)					x			x	1	1	3		2	14
Raleigh 1982			x	x		x	x			1	2	7	1	3
Sams & Pearson 1963	x					x	x				3			
Stuber 1982			x	x		x	x				1	4		7
Stuber et al. 1982			x	x		x	x				5	3		7
Swank & Phillips 1976		x				x	x				2			
Swift 1976 (spawn)	x					x		x	3		2			
Swift 1976 (toe width)	x					x	x	x			1			
Swift 1979	x					x	x	x	1	4	1			
Taylor 1982 (vegetation)		x			x			x	2	1	1			
Tennant 1975	x						x			1				
Thompson 1974	x						x				2			
Trial et al 1983 (dace)			x	x		x	x				7	6		3
Trial et al 1983 (shiner)			x	x		x	x				2	3		4
Trial et al 1983 (fall)			x	x		x	x				1	2		3
Trial et al. unpub			x			x	x				6	4		8
Waters 1976		x				x	x				2	2		
Weatherred et al. 1981		x			H		x				4			
Wesche 1980			x			x	x				1	3		
White 1976		x					x				1			
White et al. 1976					x			x		5			1	

RF = Recommends Flow Directly; HQF = produces a plot of Habitat Quality versus Flow; HQI = produces an Index of Habitat Quality at the flow measured; HSI = HQI models produced by the USFWS HEP Group; BW = Biologically Weighted input variables; SC = predicts Standing Crop (x) or hydraulic features (H) directly; C = Conceptually derived; E = Empirically derived otherwise be laborious. Such conceptually derived models are of little use until they are validated by experiment to demonstrate that the modeler's concepts were indeed valid.

Empirical or descriptive models are, for the purposes of this report, models for which the values of both input and output variables were known at the time the model was constructed. They may or may not be conceptually pleasing, but since they are derived by determing the existing relationship between input and output variables, they are inherently valid for the data set for which they were constructed. They are usually constructed in the form of a mathematical equation using linear regression techniques. Such equations are descriptive but do not imply causality since it is quite possible that the factors causing the relationship to exist are not included in the model. Therefore, empirical models also cannot be used as predictive models until they are shown to be so by experimental test.

or derived by experiment

Finally, we distinguished between measureable and unmeasurable response or output variables (identified in the first column on Table 2-2). One of the first steps in designing a model is determining whether the model output should be predictive of something that can be measured (eg. standing crop of fish), or should simply result in an index which cannot be measured by experimental test. All models that are constructed empirically must have measurable output variables, but conceptually-derived models need not.

There are two kinds of measurable response variables. One type, although measurable, is not clearly linked to the biological response variables of ultimate interest. This type is exemplified by Wetted Perimeter (Nelson 1984, Colling's 1974 rearing discharges, and White 1976) and Toe-Width (Swift 1976) methods. Both wetted perimeter and toe-width can be displayed as functions of flow, and can be measured to find out if the hydraulic simulation model used (if any) was predictive, but the question remains as to how either is linked to biological success.

The second type of measurable response variable consists of the biological variable of interest such as standing crop or riparian strip width. Models that utilize these as response variables can have their predictiveness tested directly and the model can readily be shown to be valid or invalid in a given situation. We cannot emphasize too strongly the desirability of using measurable response variables, because without them it is nearly impossible to determine if a model is producing valid results.

It is notable that only two (Binns and Eiserman 1979, Taylor 1980) of the methods that either recommend flows, or are normally used to produce output as a function of flow, use this second type of measurable response variable, and hence can be tested directly. All of the other methods that are used routinely for determining appropriate instream flows, including the U.S. Fish

TABLE 2-2 THE TYPES OF VARIABLES INCLUDED IN THE INSTREAM FLOW/HABITAT QUALITY MODELS REVIEWED

R = Recommended Flow, M = Measurable, U = Unmeasurable, C = Conceptual, E = Empirical A = arbitrary

METHOD	Type of Output Variable	Number of Input Variables	Choice of Input Variables	Transform. of Input Variables	Choice of Model Structure	Type of Parameter Estimation
Annear & Conder 1983	R	1	C		C	
Barber et al. 1980	M	10	E		A	E
Binns & Eisermann 1979	M	9	E	C	E	E
Collings 1974 (all sp.)	U	8	C		A	E
Dunham & Collotzi 1975	U	6	C	C	A	
Edwards 1983 (Buffalo)	U	11	C	C	A	
Edwards 1983 (Sucker)	U	5	C	C	A	
Edwards et al 1983 (Dace)	U	6	C	C	A	
Edwards et al 1983 (Dart)	U	8	C	C	A	
Edwards & Twomey 1982	U	12	C	C	A	
Geer 1980	R	1	C		C	
Gilbert 1984	U	12	C	C	A	
Hickman & Raleigh 1982	U	17	C	C	A	
Hoppe 1975	R	1	C		C	
Inskip 1982	U	9	C	C	A	
Larsen 1980	R	2	C		C	
Layer 1983 (all sp.)	M	14	E	E	A	E
Layher & Maughn unpub.	U	3	C	C	A	
Li et al. unpub.	M	8	C		A	E
McMahon 1982	U	20	C	C	A	
McMahon 1983	U	15	C	C	A	
Milhous et al 1984 (IFG4)	M	3	C		C	
Milhous et al 1984 (WSP)	M	4	C		C	
Milhous et al 1984 (HAB)	U	3	C	E	A	
Nelson 1984	R	1	C		C	
NGPRP 1974	R	1	C		C	
Nickelson 1976	U	3	C	C	A	
Nickelson et al 1979 (all	U	5	C	C	A	
Orsborn 1981	R	5	C		A	
Pardue & Cordes 1983	U	5	C	C	A	
Parsons et al. 1981	U	4	C		A	E
Rabern 1984 (all species)	M	21	E		A	E
Raleigh 1982	U	14	C	C	A	
Sams & Pearson 1963	R	3	C		C	
Stuber 1982	U	12	C	C	A	
Stuber et al. 1982	U	15	C	C	A	
Swank & Phillips 1976	R	2	C	C	C	
Swift 1976 (spawn)	R	5	C	C	A	E
Swift 1976 (toe width)	R	1	C		A	E
Swift 1979	R	6	C		A	E
Taylor 1982 (vegetation)	R	4	C E?		A	E
Tennant 1975	R	1	C		C	
Thompson 1974	A	2	C		C	
Trial et al 1983 (dace)	U	16	C	C	A	
Trial et al 1983 (shiner)	U	9	C	C	A	
Trial et al 1983 (fall)	U	6	C	C	A	
Trial et al. unpub	U	18	C	C	A	
Waters 1976	U	4	C	C	A	
Weatherred et al. 1981	M	4	C		C	E
Wesche 1980	U	4	C	C	C	
White 1976	R	1	C		C	
White et al. 1976	M	6	C E?		C	E

and Wildlife Service IFIM (Bovee 1982, Milhous et al. 1984), use either unmeasurable response variables or flow recommendations, the appropriateness of which cannot be tested.

Unmeasurable response variables are usually referred to by their authors as indices. (Note, however, that Binns and Eiserman's 1979 Habitat Quality Index is equal to standing crop, so that although it is referred to as an index, it is really a measurable response variable.) The most prominant examples in this category are the Habitat Suitability indices of the U.S. Fish and Wildlife Service Habitat Evaluation Procedures Group (identified as HSI models in Table 2-1). The output is dimensionless and is simply a multidimensional conceptualization that usually has not been shown by the models' authors to be correlated with anything. Others include Habitat Quality Units (Nickelson 1976), Habitat Quality Rating (Nickelson et al. 1979), Fish Habitat Index and Habitat Condition Score (Parsons et al. 1981), Relative Habitat Units (Waters 1976), and Water Resources Research Institute Cover Rating (Wesche 1980).

Falling somewhere between clearly measurable response variables such as wetted perimeter or standing crop and clearly unmeasurable ones such as the Habitat Suitability Indices, are Percent Optimum Habitat (Dunham and Collotzi 1975) and Weighted Usable Area (Bovee 1982, Milhous 1984). Percent optimum habitat is measurable in the sense that once optimum habitat is defined, the percentage of it can be measured. What makes it unmeasurable is the fact that "optimum habitat" is itself an unmeasurable concept, or at least one which cannot be tested experimentally.

Weighted Usable Area from the USFWS HABTAT model is similar to the optimum habitat response variable. Once one determines how to weight the area, it can be measured and added up as is done in the HABTAT model. The question then becomes whether the method of weighting is subject to measurement. The weighting technique used in the HABTAT model is ingenious because it substitutes behavioral habitat selection, something that is readily measured and which occurs whenever fish are present, for the variable of real biological interest, production or standing crop. This approach appears reasonable in the sense that one might expect fish to choose the optimum habitat, but on closer inspection it has several significant problems. WUA

First, fish have to pick some habitat, since they are usually not free to leave the system. If the optimum habitat is not available, either because it is not present physically or because it is defended by some other organism (usually another fish), then the selection will be of sub-optimum habitat. This probably can be circumvented experimentally by making sure all habitats and no other fish are present, but in practice is seldom addressed.

Second, a fish may be in a particular location, not because of anything at that location, but because of conditions (such as availability of hiding places) some distance away. Generally, the relationships between the location of a fish and habitat characteristics some distance away are poorly understood and and not incorporated in the weighting scheme used.

Third, optimum habitat selected behaviorally may be entirely relative. A fish may preferentially locate himself in slow water adjacent to the fastest water he can find, in order to maximize food delivery (drift). The absolute velocity may be entirely unimportant so that any weighting based on absolute velocities is misleading.

Fourth, satisfying preference for a few hydraulic variables such as depth and velocity may have no effect at all on the number of fish that can be supported in a stream. If the objective of determining appropriate instream flows is to achieve some level of biological production or to maximize it, then behaviorally weighted input variables may have little relevance to the overall decision.

Finally, it requires a large conceptual jump to conclude, as is implicit in the HABTAT model, that a large amount of barely acceptable habitat is biologically equivalent to a small amount of optimal habitat.

As long as the response variable is thought of as meaning habitat quality (or carrying capacity or some other unmeasurable quality), any test of the model that turns out to demonstrate little correlation between the response variable and the biological variable of interest, can be dismissed on the grounds that because of influences not accounted for in the model (e.g., fishing pressure), the habitat quality was not fully exploited and therefore, the test itself was invalid. This situation may be comfortable for the modeler who can take the position that the model is valid no matter what, but is not good science.

The situation can be remedied by simply converting unmeasurable response variables to measurable ones and facing up to the fact that the model may turn out to be invalid on testing and may require additional or different input terms, or may need to be replaced altogether. The conversion can be done by equating the model output with the response variable of interest (for example by using linear regression between the model response variable and some measurable variable based on a test situation), <u>then incorporating this as part of the model</u>. Wesche (1980) did that with the WRRI Cover Rating method, but as far as we know, the method was not subsequently tested on additional data sets to check for validity. For most of the other models using unmeasurable response variables, this step has been taken only as attempts at model validation (see Chapter 6), and has not subsequently found its way into the model itself. In the validation studies, the problem continues to be that failure to predict standing crop is not a demonstration of model failure.

This overall problem of model unfalsifiability is discussed in some detail with regard to the FWS HEP/HSI models by Nickum and Terrell (in press).

INPUT VARIABLES

Categories of Variables

The methods and models reviewed here use as few as 1 and as many as 21 input variables. There is evidently a clear difference of opinion among model builders as to how many are needed or desirable. We have categorized input variables as **Basin, Discharge, Hydraulic, Structural, Biological**, and **Other Physical/Chemical.** Basin variables are those specific to the basin in which the stream lies, such as drainage area, rather than to the stream itself; discharge variables are various measures of the amount of flow; hydraulic variables are the variables normally present or derived in hydraulic simulation models and include such things as depth, velocity, and wetted perimeter; structural variables are physical structural characteristics that are important biologically, usually for cover, because of their size or placement, such as percent undercut banks; biological variables, include information such as number of fish species in the drainage; and other physical/chemical variables are factors like pH, temperature, and annual rainfall. A representative categorized list of the input variables included in the models reviewed is contained in Tables 2-3a,b,c; their distribution among models in listed in Table 2-1.

The simplest solution to selection of number of input variables is that if one is trying to predict a biological response, all that is needed is the one variable closely correlated with the response of interest (if such a variable exists). The Montana method (Tennant 1975), Utah method (Geer 1980), NGPRP method (NGPRP 1974), Wetted Perimeter (Nelson 1984, Collings 1974) and Toe-Width (Swift 1976, 1979) models use the assumption (without testing it) that there is only one necessary variable. Other model builders have concluded (again, usually without testing), that a small set of variables is adequate to meet their requirements. For example, the models confined to establishing spawning flows for anadromous salmon populations are often thought of as needing only depth, velocity, and substrate as input variables, since it appears that these are the conditions fish on a spawning run need and are seeking. A good measurable response variable for this activity might be relative density of spawners using gravel meeting the model's requirements.

Other reasons to select just a few variables, all of them apparently important in the decision to include only depth, velocity and substrate type (and sometimes cover) in the FWS PHABSIM model are:

1. the variables clearly change with discharge

TABLE 2-3A. BASIN, DISCHARGE, AND HYDRAULIC INPUT VARIABLES

BASIN VARIABLES
- Basin relief
- Day of season
- Drainage area
- Drainage density
- Kilometers from ocean
- Mean basin elevation
- Mean basin length
- Mean basin slope
- Total stream length

DISCHARGE VARIABLES
- Annual flow variation
- Annual peak flow/annual minimum flow
- Annual rainfall
- Average annual flow
- Average daily flow
- Average monthly flow
- Daily flows
- Instantaneous discharge
- Median October mean flow
- Median September mean flow
- Percent flow exceedence
- Percent mean daily flow at low flow
- Seven-day, two-year low flow
- Two-year peak flood flow

HYDRAULIC VARIABLES
- Percent Area deeper than 1.5 ft.
- Depth fluctuations after spawn
- Gradient
- Manning's n
- Maximum width
- Mean column depth
- Mean column velocity
- Mean depth at 25% width-mean depth at 75% width
- Mean depth of pools in summer
- Mean pool current velocity
- Mean thalweg depth
- Mean width
- Minimum width
- Near bottom velocity
- Percent slope
- Pool volume
- Pool width/stream width
- Slope of water surface
- Stage of zero flow
- Toe-of-bank width
- Wetted area
- Wetted perimeter (WETP)

TABLE 2-3B. STRUCTURAL AND BIOLOGICAL INPUT VARIABLES

HABITAT STRUCTURE AND COVER VARIABLES
- Area overhanging vegetation
- Area veg. overhanging <1 meter
- Area veg. overhanging 1-2 meters
- Bank locations with brush
- Bank locations with trees
- Forested area
- Frequency of logs and boulders within 50 cm upstream
- Frequency of turbulance cover at depths > 5 cm
- Frequency overhanging cover within 50 cm upstream, depth > 5cm
- Frequency rootwads within 50 cm upstream, depth > 5cm
- Frequency undercut banks within 50 cm upstream, depth > 5cm
- Frequency undercut boulders within 50 cm upstream, depth > 5cm
- Length overhead bank cover
- Number of rocks/reach
- Percent boulder & log cover
- Percent brush cover
- Percent cover
- Percent eroding banks
- Percent first-class pools
- Percent instream cover
- Percent instream bank vegetation
- Percent overhead cover
- Percent pools
- Percent pools with canopy
- Percent stablized banks
- Percent vegetative canopy
- Percent 1-3 inch gravel
- Total cover area

BIOLOGICAL VARIABLES
- Annual # of frost free days
- Bank locations with brush
- Bank locations with trees
- Food item density (#/area)
- Food item diversity
- Invertebrate drift density
- Number of fish species in drainage
- Percent shade between 1000-1400 hrs
- Percent vegetation on bank
- Rel. distance from center of range
- Submerged vegetation density
- 2 * % decid. trees + % grass + % conifers
- 2 * % shrubs + 1.5 * % grass + % trees

TABLE 2-3C. OTHER PHYSICAL AND CHEMICAL INPUT VARIABLES

OTHER PHYSICAL VARIABLES

Temperature

- Max temp,. upstream migration
- Max temp., embryo develop
- Max. temp., downstream migration
- Mean annual air temperature
- Mean annual temperature
- Mean max summer temperature
- Mean summer temperature
- Mean temp.,embryo develop.
- Mean winter temperature

Substrate Size

- Actual size distributions
- Arbitrary scale
- Arithmetic mean (.3-8cm)
- Dominant (>50%) size
- log-10 avail. spawning area
- Percent <3mm diameter during spawning
- Percent <3mm in riffle-run areas
- Percent <6mm in diameter
- Percent Embeddedness
- Percent fine substrate
- Percent 10-250mm in diameter
- Percent 10-40mm in diameter

Chemical variables

- Alkalinity
- Annual DO range
- Biochem. Oxygen Demand
- Color
- Max turbidity in summer
- Maximum (or minimum) pH
- Mean annual minumum DO
- Mean annual pH
- Mean annual turbidity
- Mean TDS May-Oct
- Min DO during embryo develop.
- Min DO during embryo develop.
- Min DO during low water
- Min DO upstream migration
- Minimum DO downstream migration
- Minimum DO during rearing
- Nitrate nitrogen
- Specific conductivity
- Total hardness
- Total organic carbon
- Total Phosphorous

2. they are readily measured and amenable to simulation modeling

3. the current version of the computer portion of the model will accept only a limited number of input variables

In the three empirical models reviewed where large numbers of variables were considered, the initial set was chosen because of availability (Binns and Eiserman 1979, Layer 1983, Rabern 1984), and was decreased using statistical criteria to produce a final model.

In Binns and Eiserman's (1979) model, the initial set of conceptually chosen candidate data was tested for linear correlation with the response variable and only those variables somewhat correlated were retained. This was also one of several variable selection procedures used by Rabern (1984). Binns and Eiserman subsequently created new variables by multiplying some of the existing ones together in order to improve model performances.

Taylor (1980) correctly rejected some candidate variables because they were strongly correlated with one another, then added new variables when the original model failed to perform adequately, another effective approach.

In addition to categorizing models by characteristics of their response variables, we have also categorized them by the way the input variables were chosen and treated within the model.

All models were divided into those for which the values of the variables are weighted to reflect their biological suitability prior to being used in the model, and those in which the variables are use in their unweighted state. Those that use biological transformation are identified under the BW heading on Table 2-1. The reason for performing such a weighting is to transform the input data so that it is linearly correlated to the output variable. This process is described in detail in Chapter 3 on transforming variables.

Finally, in Table 2-2 we have indicated whether the choice of input variables, was conceptual or empirical; whether the input variables were transformed biologically and if so whether it was done conceptually or empirically, whether the choice of the model structure was conceptual, empirical, or arbitrary, and finally whether parameter estimation was done. All of these features of model building are discussed, as they relate to instream flow method, in the next chapters.

Tables 2-1 and 2-2 show some interesting patterns. Less than a third of the models were developed empirically, and approximately half of them use biologically weighted input variables. The total number of input variables range from 1 to 21 per model, with hydraulic, structural and physical/chemical variables most prominent, hydraulic variables appear in all but 10 of the methods and structural variables appear in all but 21.

The selection of input variables was conceptual in all but 4 of the models. That is, the authors decided which variables to include because they thought they understood what variables were important. For those variables which were transformed prior to inclusion in the model, most were transformed based on generalized considerations found in the literature and we have called that conceptual. The authors of only two methods, Layer's and the IFIM HABTAT model advocated using empirical field data to make the transformations.

Model structure was considered to be arbitrary if we could see no mechanistic reason for the way the input variable terms were combined, and conceptual if there was a mechanistic approach to the aggregation of terms. Only one method, that of Binns and Eiserman appeared to arrive at a model structure empirically by trying out several structures to see which worked best.

Finally in some models the various input variables were weighted as to their importance. In other words they had associated with them coefficients or parameters. In all these cases the parameter estimation was empirical, usually through the process of multiple linear regression modeling. In most others all terms were considered essentially equal.

Figure 2-1 summarizes the scheme used for categorizing the various methods. Figure 2-2 shows the way in which the various methods are arrayed on a two dimensional axis representing choice and transformation of variables. Figure 2-3 shows the way the models arrayed on a two-dimensional axis representing choice of overall model structure and the method of parameter estimation.

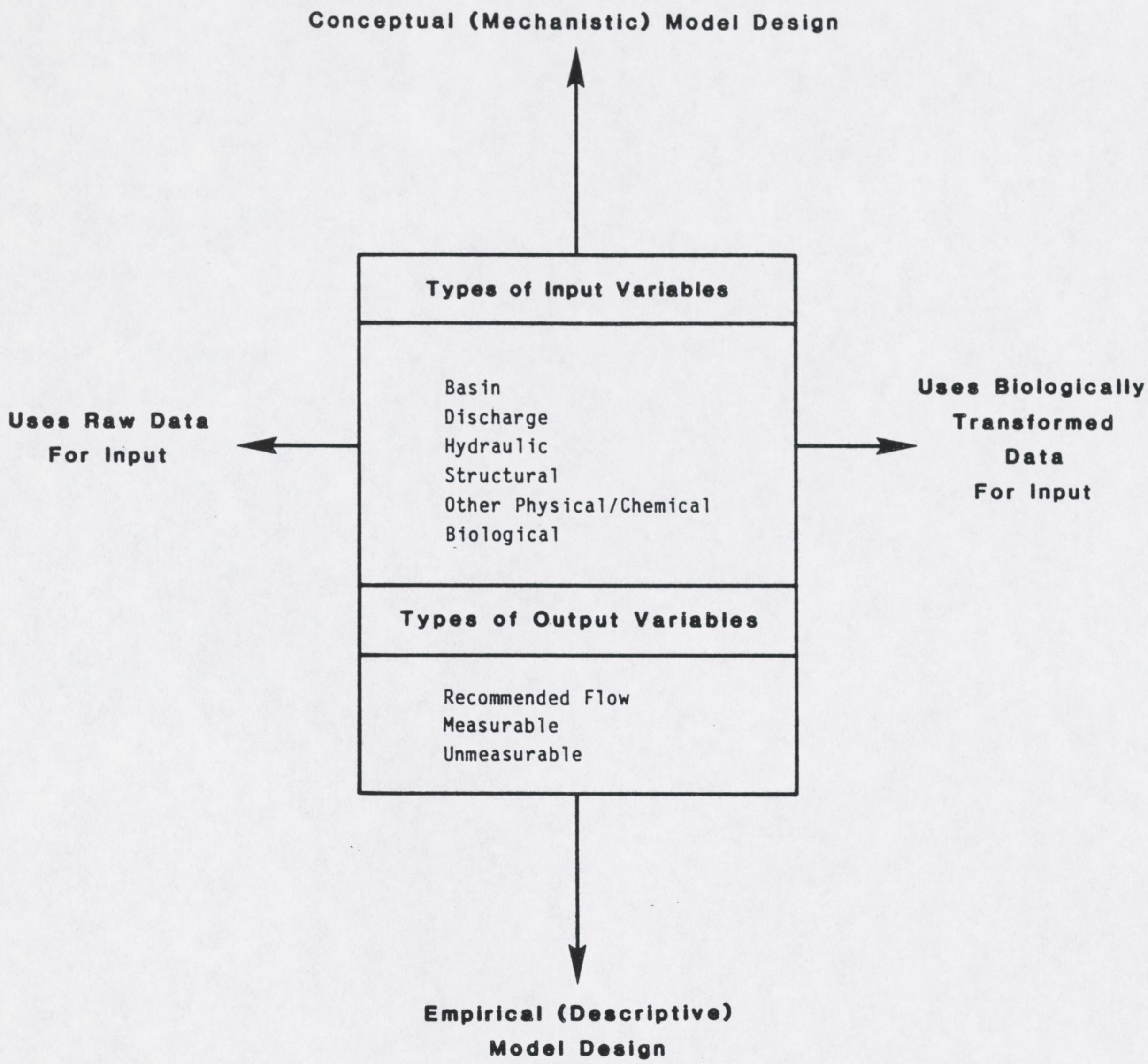

Figure 2-1. Scheme used for categorizing methods and models reviewed in this report.

Choice and Transformation of Variables

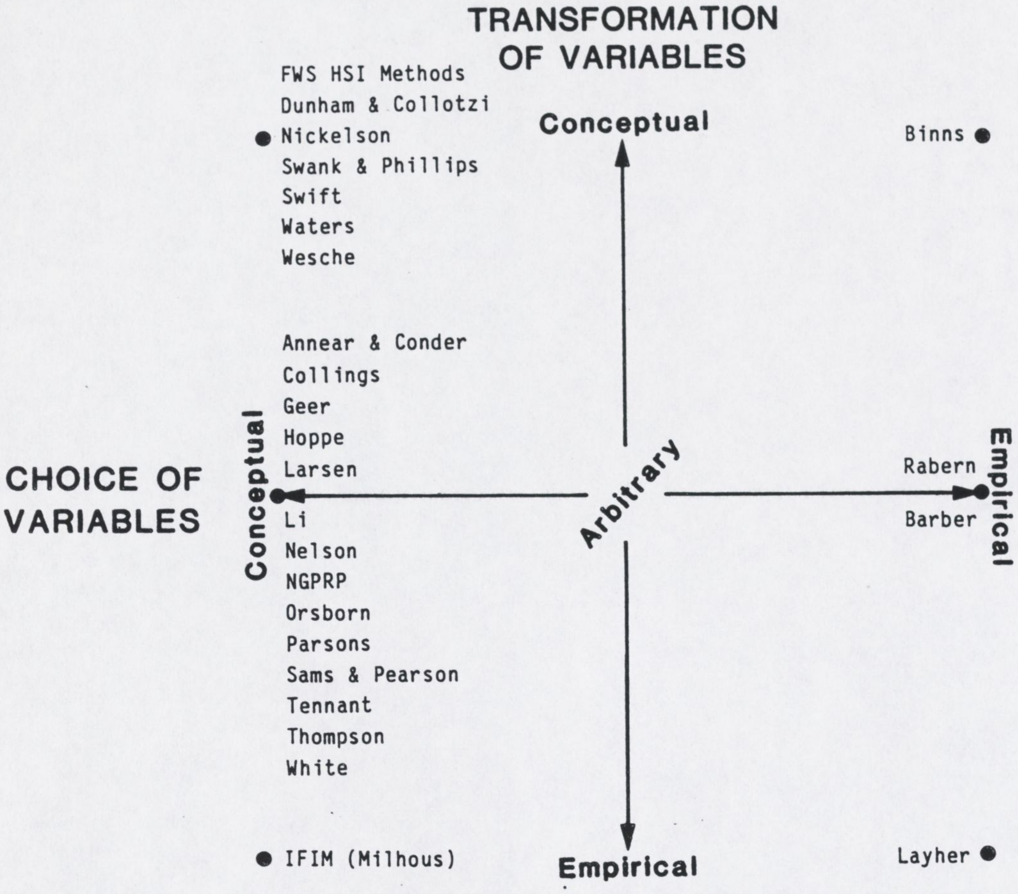

Figure 2-2. This diagram shows that both choice of variables and transformation of them ranges from conceptual to empirical. The distinction between conceptual and empirical in choice of input variables is whether or not some were picked or discarded based on their correlations with the output variable or a surrogate output variable. Input variables were considered arbitrarily transformed if no transformation was done; conceptually transformed if not based on the modeler's experiments, and empirically transformed if based on the modeler's data.

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Choice of Model Structure and Parameter Estimation

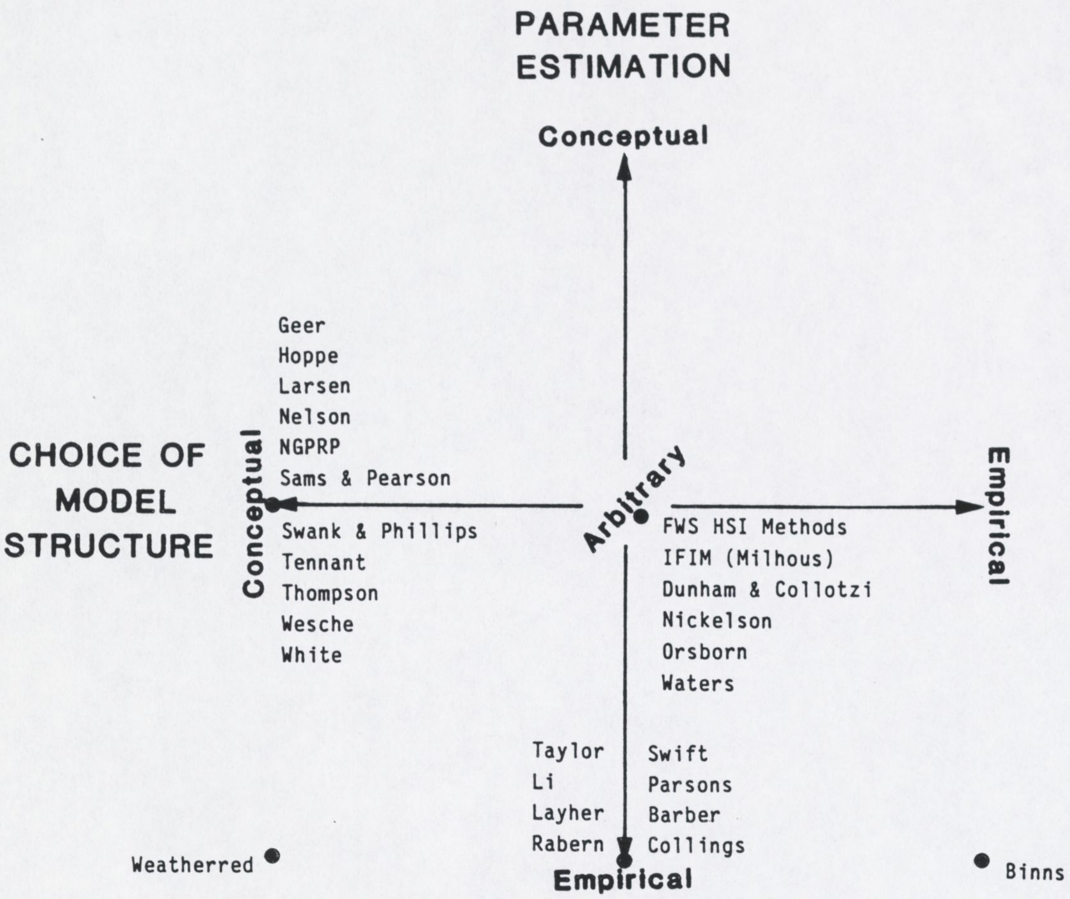

Figure 2-3. This diagram shows that the choice of model structure and the estimation of parameters can range from conceptual to empirical, and shows the extent of such variation in some of the existing methods. The choice of model structure was called conceptual if it was based on some mechanistic principal envisioned by the author; arbitrary if it simply involved multiplying all variables together or some variant thereof, and empirical if several model structures were tried and the most predictive chosen.

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CHAPTER 3: TRANSFORMING VARIABLES

One of the most prominant aspects of instream flow models is the transformation of data for physical variables into biological suitability indices. The reason for conducting this type of data transformation is to linearize physical variables with respect to their biological significance. This Chapter describes some of the existing approaches to making these transformations and their limitations.

WHAT IS DATA TRANSFORMATION?

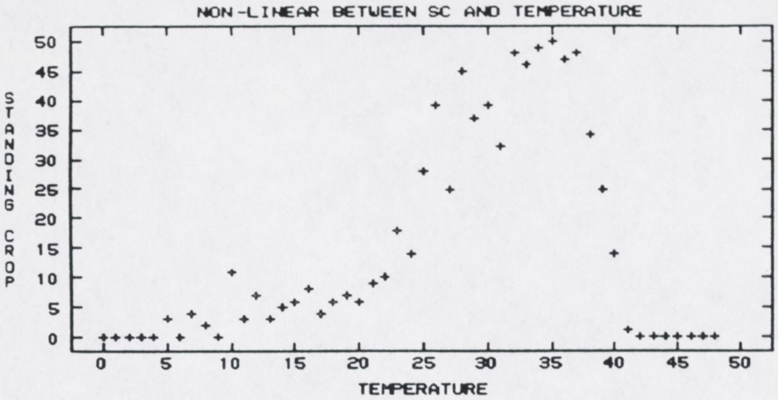
Data transformation is a process which systematically alters the numerical value of each data point, ideally, in a reversible way without decreasing the information content of the set of data. It can be done using recognizable mathematical functions, for example taking the logarithm of each data point to generate a new data set which is then said to be log transformed. A large variety of standard numerical transformations of this type is in common use (including taking the square root, the inverse, the cosine, etc.) of each member of the original data set to produce new transformed data set. The usual reason is to cause the transformed data to be more linearly related to some response variable than were the raw data, thus allowing a better fit using linear regression techniques. In other words, if when a particular set of input data such as velocity measurements is plotted against a response variable such as standing crop, it describes a curved relationship rather than a straight line, then linear correlation and regression techniques will show a poor fit even though there may be a strong functional relationship between the input and response variables. Any systematic transformation of the values of either input or response variables or both that increases in linearity of the relationship is desireable because it will facilitate analysis using linear regression techniques.

BIOLOGICAL DATA TRANSFORMATION

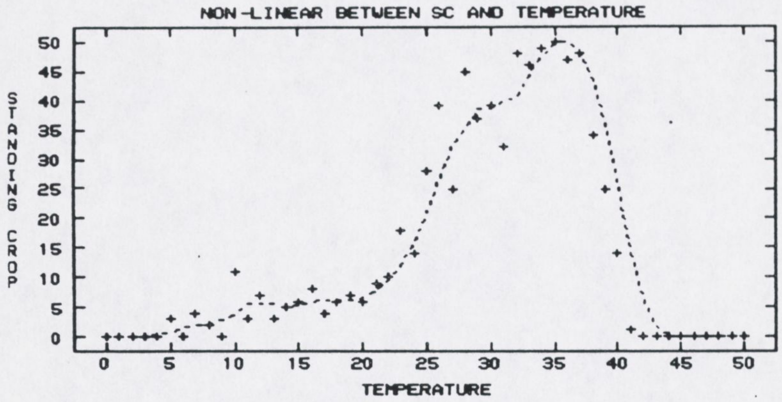
Biological response variables often cannot be linearized satisfactorily using a standard mathematical function, yet they may still have a strong relationship to a particular input variable. The proper approach used in biological transformation is to discover this relationship empirically and use it to transform the input variable. This process is illustrated in Figure 3-1.

Figure 3-1a is a scatter diagram of a hypothetical data set showing the effects of water temperature on standing crop. There is clearly a relationship between the two variables, but it is not known how to transform the temperature data so that it will be linear with respect to standing crop. Consequently, the relationship is described empirically by fitting a curve to it (Figure 3-1b), in this case by using a 3-point running mean, but the curve could have been fit by some other means or even by eye.

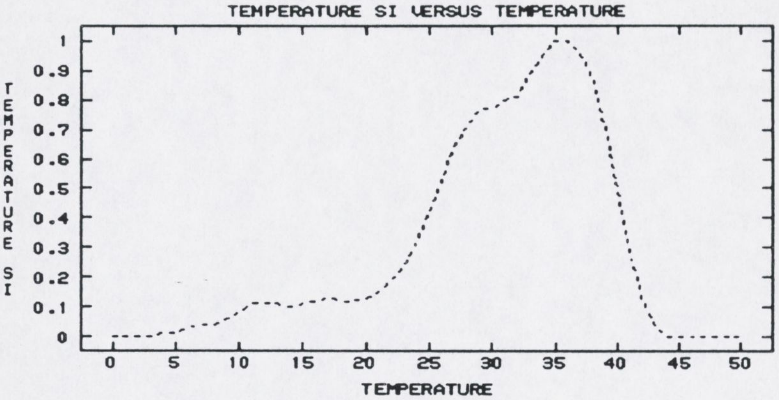
A. The raw data show a clearly non-linear relationship between temperature and standing crop.



B. A curve is fit to the non-linear relationship, in this case by using normalized running means.



C. The curve is converted to a suitability index by normalizing it to 1.0



D. This plot shows that standing crop is far more closely linearly related to the temperature suitability index than to the raw temperature data.

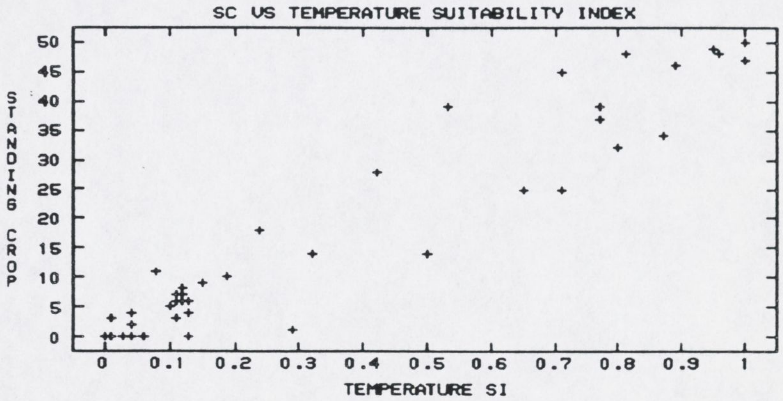


Figure 3-1. The process of empirically transforming physical input variables into suitability indexes that are linearly related to the biological output variable.

Once the values of the curve are known they can be normalized to a dimensionless suitability index ranging from 0-1 by dividing all values by the maximum value (Figure 3-1c). The standing crop data can then be plotted against not temperature as they were in Figure 3-1a, but against the suitability index corresponding to the temperature (found by using the relationship shown in Figure 3-1c) and the resulting plot (Figure 3-1d) will be much more linear than the original data. The temperature suitability index rather than temperature would then be used in any subsequent linear regression model or linear equation describing the quality of the habitat.

HABITAT SUITABILITY INDEX BIOLOGICAL DATA TRANSFORMATION

The simplest approach to biological data transformation is the one used in most of the USFWS HSI models in which the basis for the shape of the transformation curves are observations taken from the literature. Figure 3-2 a, b, c is a set of these curves taken from the HSI Brown Trout Model (Raleigh 1982) and is reproduced here to illustrate the fact that curves of a variety of shapes are possible, some linear, some non-linear and some step functions, and since the specific shape of the curves is conceptual, the specific shape of each of the curves is more closely related to the opinions of the modeller drawing them than to any empirical data. It is far preferable, however, to prepare curves from a set of real data than by drawing conceptual lines.

DEALING WITH DATA SCATTER

Data suitable for making transformation curves rarely comes in as unscattered a form as shown in Figure 3-1. It is more common for scatter diagrams of a biological response variable plotted against various input variables to have a great deal of scatter. Figure 3-3 shows a typical data set of scatter diagrams of biomass plotted against the 11 input variables used in the FWS HSI model for cutthroat trout. For most of the variables there is little if any relationship, and consequently, empirical transformation of the type shown in Figure 3-1 would not result in a relationship much more linear than the untransformed data. Nor does such a transformation in any way reduce the scatter of the data.

Kellogg & ? 1983 in refs

To resolve this problem several different techniques have been used and three of these are illustrated (using the tabular data from Kellog et al. 1955) in Figure 3-4. Figure 3-4a shows the approach used by Li et al. (unpublished) in their attempt to produce predictive suitability index curves for cutthroat trout and coho salmon in Oregon. They simply drew a piece-wise linear envelope around their scatter diagrams and justified it on the grounds that only those points at the periphery of the envelope were valid for modelling habitat because all the other points represented samples from streams not at carrying capacity.

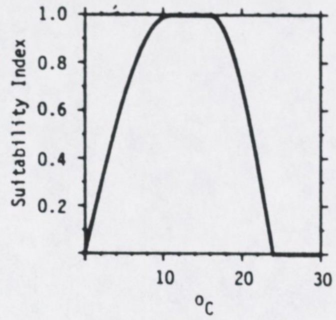
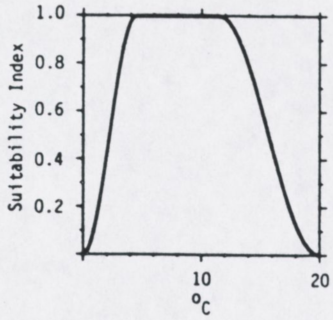
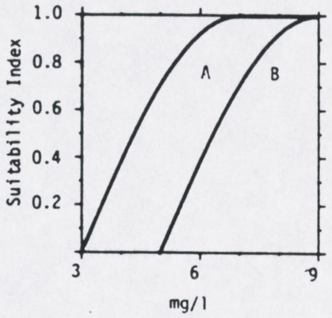
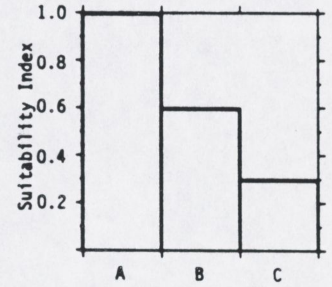
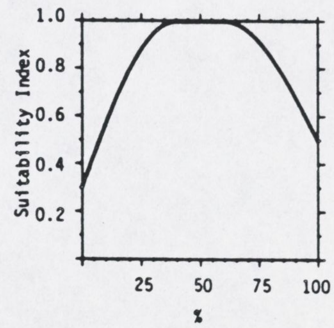
Rational for Curve	Description	Curve
Average maximum daily temperatures have a greater effect on trout growth and survival than minimum temperature.	Average maximum water temperature (°C) during the warmest period of the year (adult, juvenile, and fry). For lacustrine habitats, use temperature strata nearest optimum in dissolved oxygen zones of >3 mg/l.	
The average maximum daily water temperature during embryo development related to the highest survival of embryos and normal development is optimum.	Average maximum water temperature (°C) during embryo development.	
The average minimum daily dissolved oxygen level during embryo development and the late growing season that is related to the greatest growth and survival of brook trout and trout embryos is optimum. Levels that reduce survival and growth are suboptimum.	Average minimum dissolved oxygen (mg/l) during the late growing season low water period and during embryo development (adult, juvenile, fry, and embryo). For lacustrine habitats, use the dissolved oxygen readings in temperature zones nearest to optimum where dissolved oxygen is >3 mb/l. A = $\leq$ 15° C B = $\geq$ 15° C	
The dominant substrate type containing the greatest numbers of aquatic insects is assumed to be optimum for insect production.	Dominant ($\geq$ 50%) substrate type in riffle-run areas for food production. A) Rubble or small boulders or aquatic vegetation in spring areas dominant, with limited amounts of gravel, large boulders, or bedrock. B) Rubble, gravel, boulders, and fines occur in approximately equal amounts or gravel is dominant. Aquatic vegetation may or may not be present. C) Fines, bedrock, or large boulders are dominant. Rubble and gravel are insignificant ($\leq$25%).	
The percent pools during late summer low flows that is associated with the greatest trout abundance is optimum.	Percent pools during the late growing season low water period.	

Figure 3-2a. Typical biological transformation curves from a USFWS Habitat Suitability Model showing the limited amount of data used to establish the shapes of the curves (from Raleigh 1982).

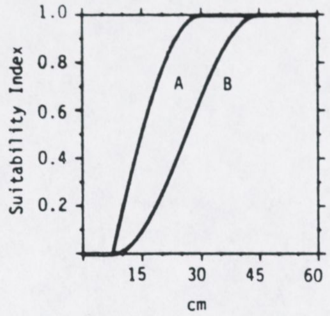
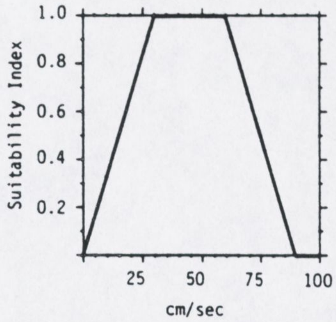
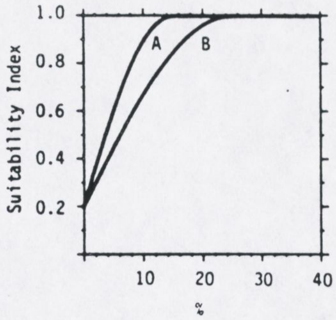
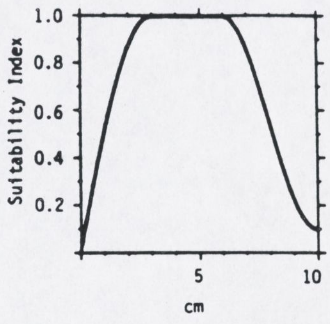
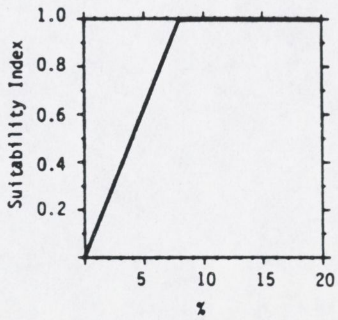
Rational for Curve	Description	Curve
The average thalweg depths that provide the best combination of pools, instream cover, and instream movement of adult trout is optimum.	Average thalweg depth (cm) during the late growing season low water period. A = stream width $\leq$ 5 m B = stream width $\geq$ 5 m	
The average velocity over the spawning areas affects the dissolved oxygen concentration and the manner in which waste products are removed from the developing embryos. Average velocities that result in the highest survival of embryos are optimum. Velocities that result in reduced survival are suboptimum.	Average velocity (cm/sec) over spawning areas during embryo development.	
Trout standing crops are correlated with the amount of usable cover present. Usable cover is associated with water $\geq$ 15 cm deep and velocities $\leq$ 15 cm/sec. These conditions are associated more with pool than riffle conditions. The best ratio of habitat conditions is about 50% pool to 50% riffle areas. Not all of a pool's area provides usable cover. Thus, it is assumed that optimum cover conditions for trout streams are reached at <50% of the total area.	Percent instream cover during the late growing season low water period at depths $\geq$ 15 cm and velocities < 15 cm/sec. A = Juveniles B = Adults	
The average size of spawning gravel that is correlated with the best water exchange rates, proper redd construction, and highest fry survival is assumed to be optimum for average-sized brook trout. The percentage of total spawning area needed to support a good trout population was calculated from the following assumptions:	Average size of substrate between 0.3-8 cm diameter in spawning areas, preferably during the spawning period. To derive an average value for use with graph V_7, include areas containing the best spawning substrate sampled until all potential spawning sites are included or the sample contains an area equal to 5% of the total brook trout habitat being evaluated.	
The substrate size range selected for escape and winter cover by brook trout fry and small juveniles is assumed to be optimum.	Percent substrate size class (10-40 cm) used for winter and escape cover by fry and small juveniles.	

Figure 3-2b.

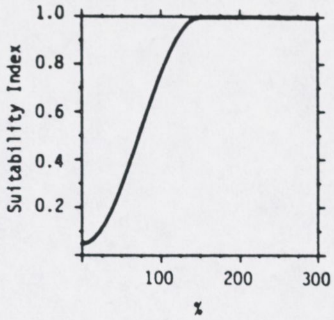
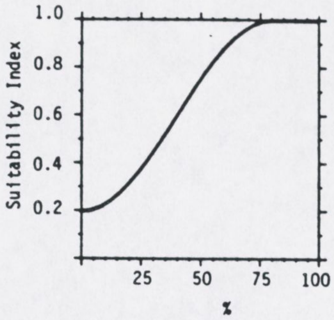
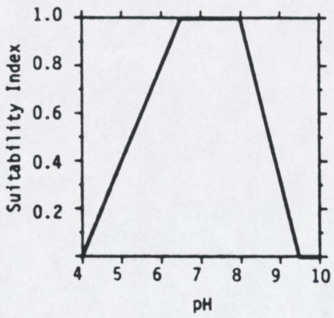
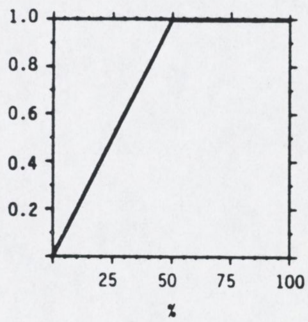
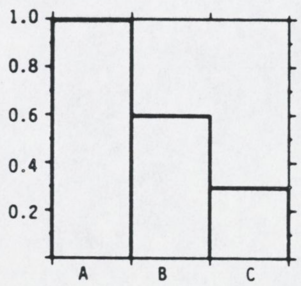
Rational for Curve	Description	Curve
The average percent vegetation along the streambank is related to the amount of allochthonous materials deposited annually in the stream. Shrubs are the best source of allochthanous materials, followed by grasses and forbs, and then trees. The vegetational index is a reasonable approximation of optimum and suboptimum conditions for most trout stream habitats.	Average percent vegetation (trees, shrubs, and grasses-forbs) along the streambank during the summer for allochthonous input. Vegetation Index = 2 (% shrubs) + 1.5 (% grasses) + (% trees) + 0 (% background). (For streams ≤ 50 m wide).	
The average percent rooted vegetation and rocky ground cover that provides adequate erosion control to the stream is optimum.	Average percent rooted vegetation and stable rocky ground cover along the streambank during the summer (erosion control).	
The average annual maximum or minimum pH levels related to high survival of trout are optimum.	Annual maximal or minimal pH. Use the measurement with the lowest SI value. For lacustrine habitats, measure pH in the zone with the best combination of dissolved oxygen and temperature.	
Flow variations affect the amount and quality of pools, instream cover, and water quality. Average annual base flows associated with the highest standing crops are optimum.	Average annual base flow regime during the late summer or winter low flow period as a percent of the average annual daily flow.	
Pool classes associated with the highest standing crops of trout are optimum.	Pool class rating during the late growing season low flow period (Aug-Oct). The rating is based on the percent of the area containing pools of the three classes described below. A) ≥30% of the area is comprised of first-class pools. B) ≥10% but <30% first-class pools or ≥50% second-class pools. C) <10% first-class pools and <50% second-class pools.	

Figure 3-2c

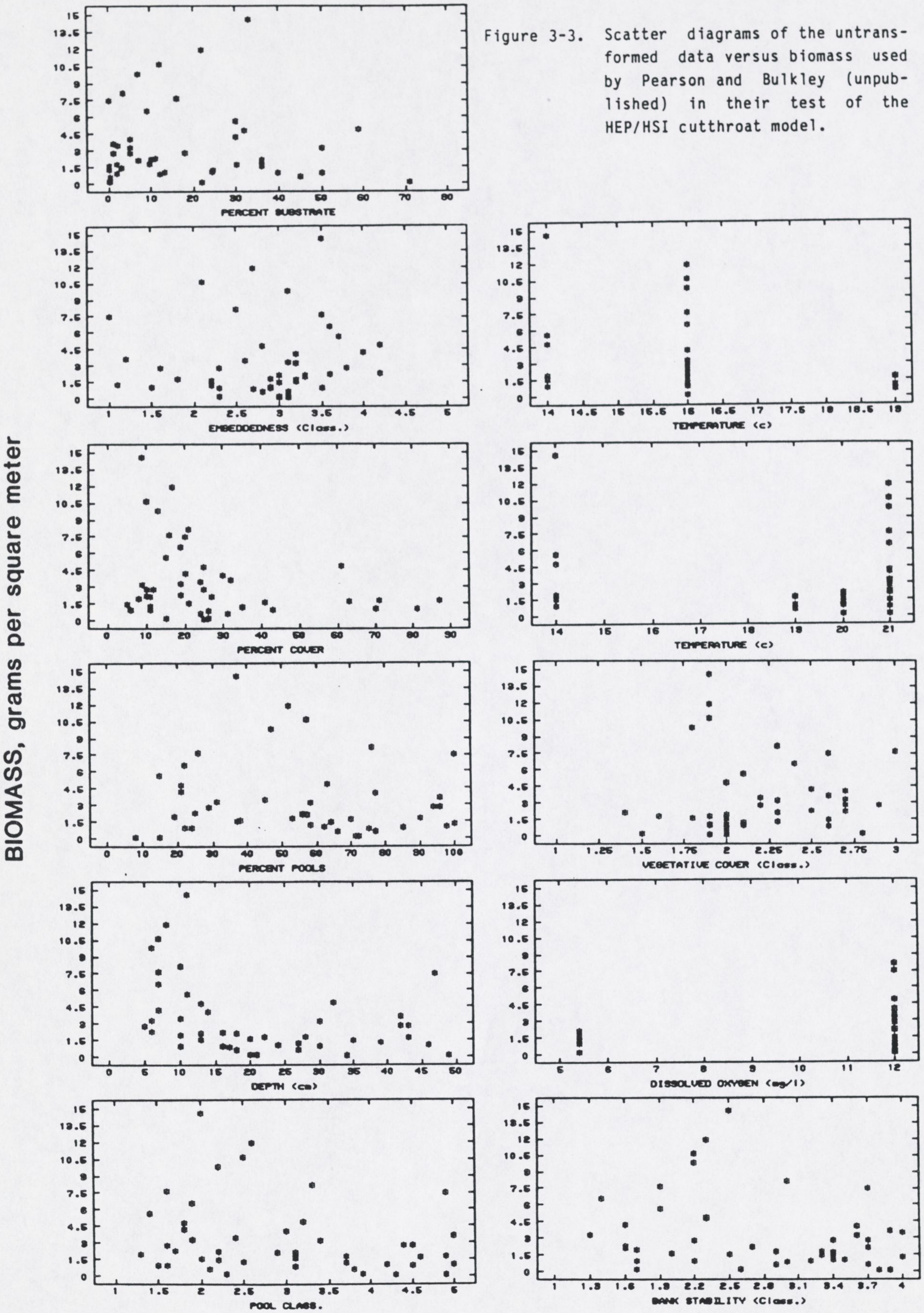


Figure 3-3. Scatter diagrams of the untransformed data versus biomass used by Pearson and Bulkley (unpublished) in their test of the HEP/HSI cutthroat model.

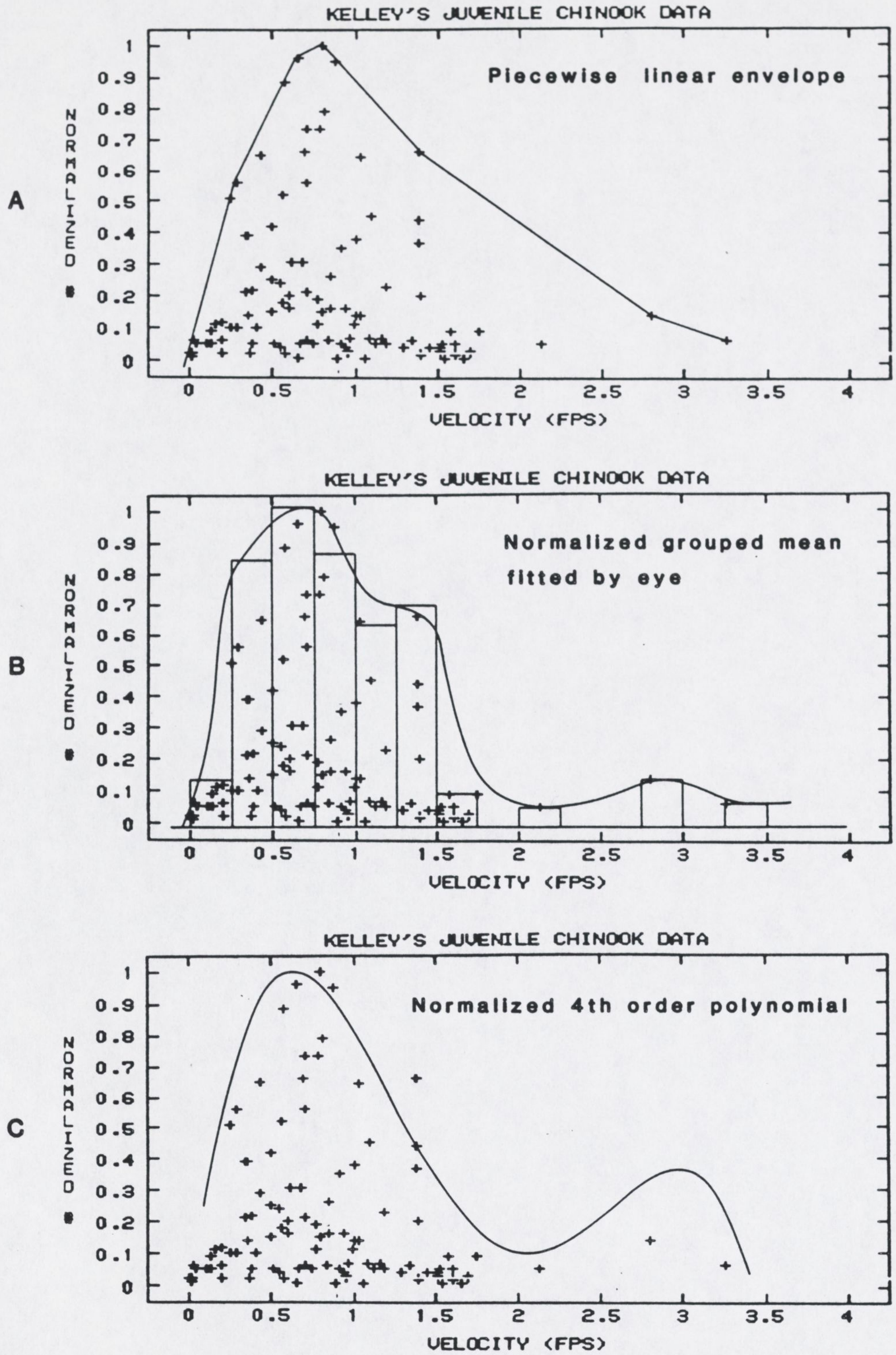


Figure 3-4. Three methods of fitting biological transformation curves (suitability indexes) to standing crop versus velocity data. The top figure shows the drawing of an envelope in the manner used by Li et al. (unpublished). The middle figure shows the technique used by Layher (1983), grouping the data into bins, taking the mean of the bins, then normalizing the means and drawing a line through the means by eye. The bottom illustration shows the technique of fitting a 4th order polynomial regression, as was done by Kelley et al. (1985) then normalized to 1.0. The raw data are from Kelley et al. (1985) and were normalized by us for this illustration.

Kellogg and Gift 1983 ?

Figure 3-4b demonstrates the technique used by Orth and Maughn (1982) and by Layer (1983) in the development of empirical HSI models. The technique used is to segregate the data by velocity categories (in this case we used 0.25 fps increments) and calculate the mean value of each grouping. The groupings are then normalized to 1.0, and a line fitted to them by eye.

Figure 3-4c shows the technique used by Kelley et al. (1985) on these data. They fitted a polynomial (we used a fourth order polynomial in this example) and then normalized the peak of the polynomial to 1.0 to form an SI curve.

All of these techniques result in curves that resemble one another, but differ in particulars. The only reproducible technique in this illustration, however, is the fitting of a polynomial. The other two approaches depend on judgement when fitting the curve by eye and result in different curves when done by different people. On some data sets quite substantially different curves could be drawn, and it is common for authors using the technique shown in Figure 3-4b to ignore large peaks if they do not correspond to the authors conceptual image, and to add peaks for the same reason. Figure 3-5 shows a sample of this type of curve fitting from Layer (1983) illustrating the variability of fits common to this technique.

PRODUCING SI CURVES FROM FREQUENCY HISTOGRAMS

The other type of data that has generally been used for producing SI curves is frequency histogram data, usually based on behavioral observations. This is the technique used in producing probability-of-use curves (now called SI curves) for use with the FWS HABTAT computer model. These data are collected in such a way that a scatter diagram of the type shown in Figure 3-4 does not occur. Rather, a series of observations of fish is made and the depth and velocity (and other features) at the location of each observation is recorded. In order to convert these data to a two dimensional plot for conversion to an SI curve the only possibility is to plot frequency of observation since each data point consists of a single observation. In producing such a histogram a decision must first be made as to how to combine the data along the x-axis. If all the data were combined into a single bin, there would simply be a rectangularly shaped graph showing no graded level of suitability. If the data are not combined together into bins at all, there would be a straight line parallel to the horizontal axis. Consequently, the bin size used must be a purposful decision, which, it turns out, has consequences for the shape of the resulting SI curve. Figure 3-6 shows a typical depth-frequency histogram data set analyzed using a range of bin size groupings from 0.2 feet to 1.0 ft deep. Notice that the highest resolution (0.2 ft) has several bins which contain little or no data scattered throughout the set but that as bin size is increased these empty bins disappear, resulting in a smoothing of a line drawn between bin mid-points.

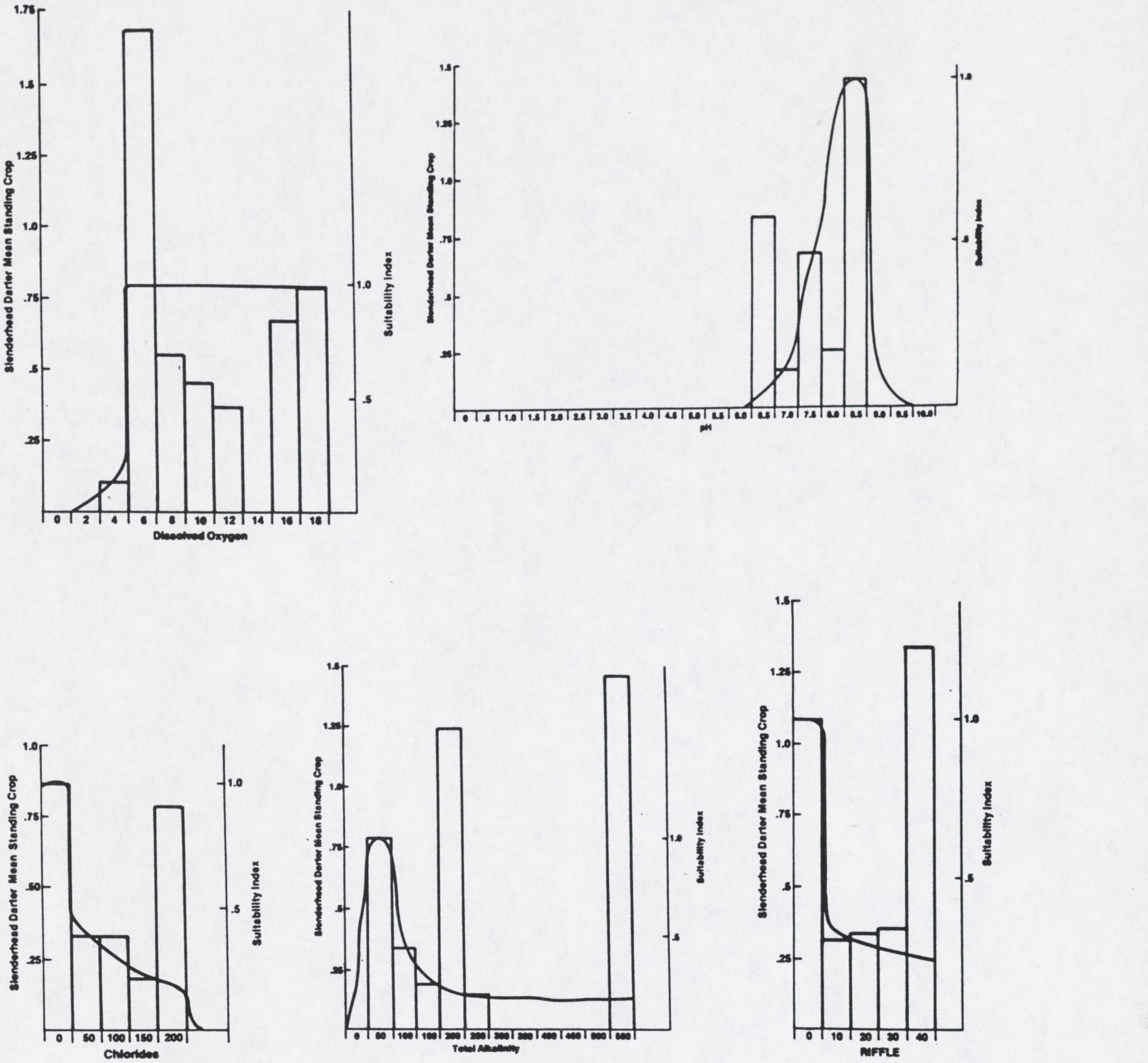


Figure 3-5. Some examples of inconsistencies common to SI curves fit to histograms by eye. Note that peaks occurring where the investigator disagreed with their presence were ignored in fitting the continuous function (line) which was then used as the suitability index. (From Layher 1983).

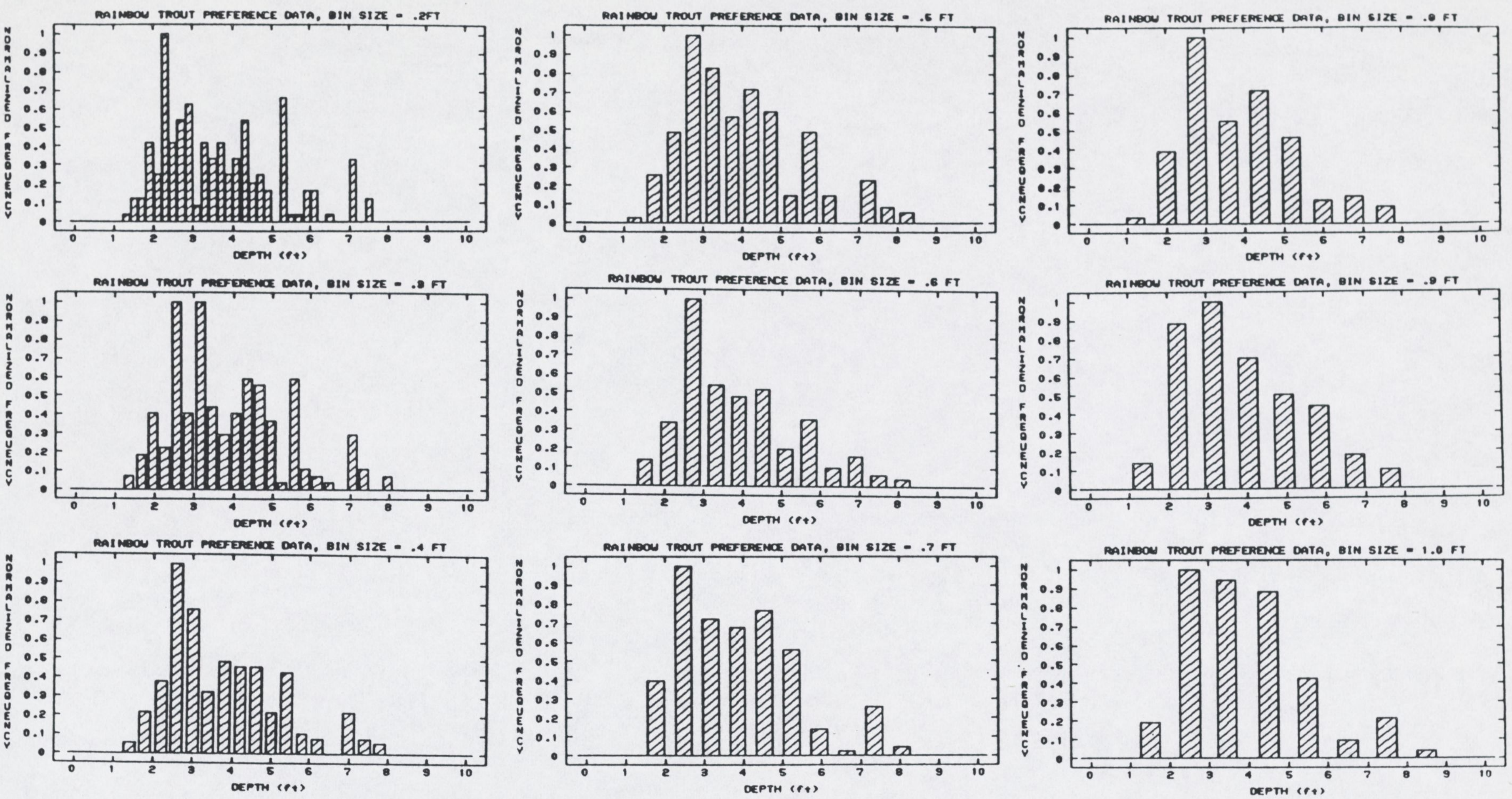


Figure 3-6. EFFECT OF BIN (INTERVAL) SIZE SELECTION ON PREFERENCE* CURVE SHAPE. The top 9 figures were all generated using the same set of observations of the depths at which 260 adult rainbow trout were sighted. In order to demonstrate the effect of bin size selection on the resulting preference curves, the bottom left figure shows the lines resulting from connecting midpoints of adjacent bins, and the bottom right figure shows the result of one individual drawing envelopes around each histogram by hand. Both techniques result in a wide range of preferences at most depths.

*These data represent direct sightings, uncorrected for habitat availability.

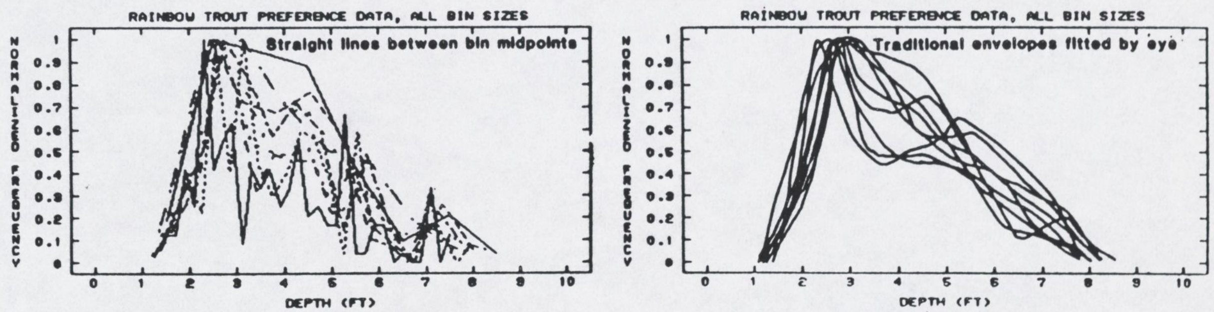


Figure 3-6 includes a graph showing the different curves that result if straight lines are drawn between the bin mid-points of each of the histograms shown. There is a very large difference in the resulting SI curve, depending on which bin size is used. Similarly, if curves are fit by eye to each of the histograms (smoothing out the artifacts introduced by empty bins) there is also a substantial difference between curves produced using various bin sizes.

This difference in curve shape, particularly in the steep parts of the curve can make a very substantial difference in the predicted amount of habitat and hence, instream flow recommendation resulting from a model using the curve. It is important to realize, however, that there is no theoretical reason to use one bin size over another. All of them produce equally valid results, but with decreasing information content as bin size is increased.

A related phenomenon is the effect of beginning the data combining process at different values. If, for example, it is decided to combine velocity data in such a way that 3 velocity units are combined in each bin (e.g. velocities were measured to the nearest 0.05 ft per second but bin size is established at 0.15 ft per second) then the point where the combining process starts can have considerable effect on the shape of the curve. Figure 3-7 shows the three possible histograms resulting from such an exercise when the binning process starts on the lowest value (top illustration), the second to the lowest value (middle illustration), and the third from the lowest value (bottom illustration). The three histograms have quite different shapes and none is more correct or accurate than the others (except that for the purposes of the illustration we simply eliminated the lowest and second lowest values). Baldridge and Amos (1981) did a similar exercise and argued that the correct grouping was one which resulted in a monotonic function (one with no dips in it) and minimized variability between adjacent bins, but concluded that since no single grouping necessarily fit these criteria, ultimately a biologist had to determine which grouping to use. They did not specify what criteria the biologist should use, however, and the reality is that there are no theoretically correct criteria.

CONVERTING HISTOGRAMS TO CONTINUOUS FUNCTIONS

Once the histogram has been selected and formalized it is then usually desireable to fit a continuous function to it for use as an SI curve. This can be done using the techniques illustrated in Figure 3-1 and 3-4, or variations of them. A technique we have found useful for smoothing and eliminating dips is to apply a succession of 3 point running means as illustrated in Figure 3-8. Each time a running mean is applied the result is normalized, and if another pass is desired, it is made on the normalized results of the previous pass. In the example shown four

*

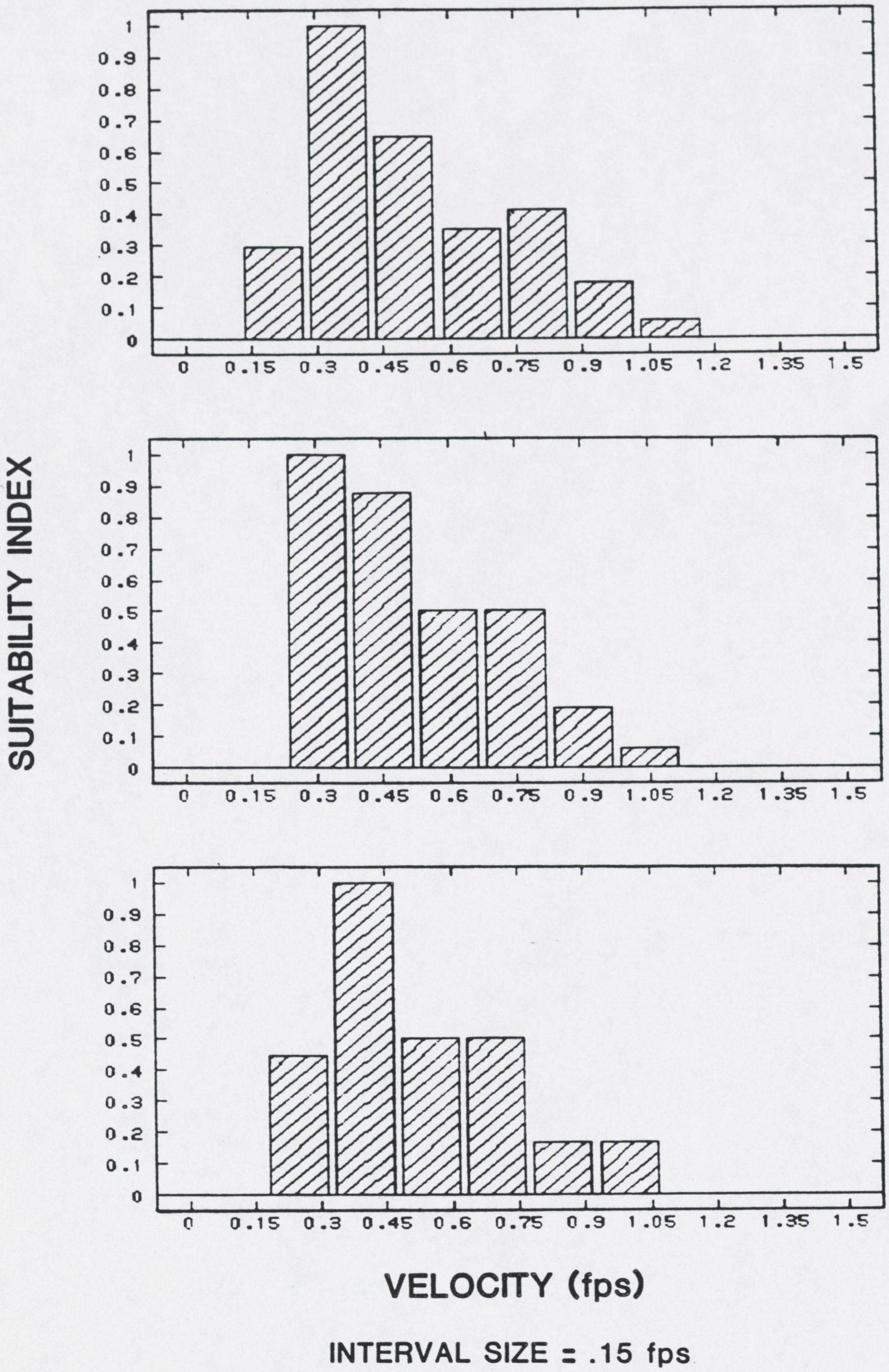


Figure 3-7. An example of the effect on the shape of frequency histograms of beginning at different places in the data set when grouping the data.

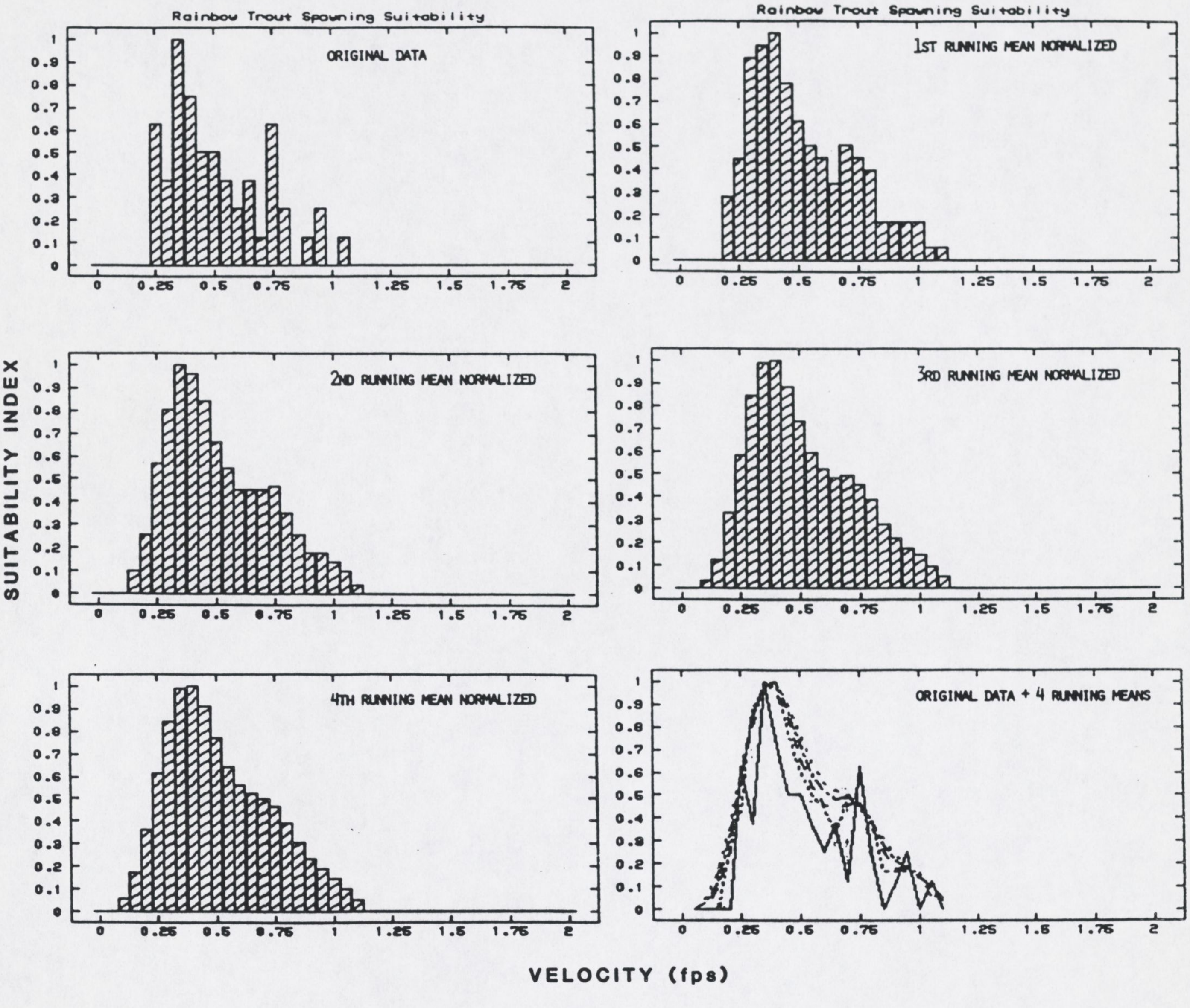


Figure 3-8. Illustration of the effect of using 3 point running means followed by normalization to smooth uneven histogram data into a biological transformation curve. The procedure was carried out 4 successive times to eliminate dips. (Data collected by EA).

successive running means were performed to meet the arbitrary criterion of achieving monotonicity (lack of dips). The continuous functions that resulted from connecting the midpoints of the successive running mean histograms are shown in Figure 3-8 as well.

Another approach to fitting a continuous function to the histogram data is illustrated in Figure 3-9 in which a second order polynomial (quadratic) function was fit to the same data used in Figure 3-8. For data set 1 (Figure 3-9) no leading or trailing zero values were included in the data set and the resulting curve was nearly linear. For data set 2 a single zero value was added at zero velocity to force the function toward zero suitability at zero flow. In data set 3, leading and trailing zeros were added to further constrain the function. For this particular histogram a second-order polynomial does not produce a very satisfactory fit. A third-order polynomial was then fit and normalized (Figure 3-10) using the same approach with better results, but still not a particularly good fit. In Figure 3-4c the results of using a 4th order polynomial are shown.

Generally we have found that better fits are achieved using running means than using polynomials although other authors have used alternative polynomial fitting technique with good results. Gore and Judy (1981), for example, fit a 4th-order polynomial to the cumulative frequency distribution of their histogram data, than took the first deriative of the polynomial and normalized it thereby creating an SI curve. They also fit parabolic functions to some of their data. Voos (1981) fit exponential polynomial probability density functions to the data of Prewitt (1980) but the data were quite scattered and it is difficult to judge the benefits of this process.

A final curve fitting approach that may occasionally have merit is to superimpose a continuous distribution, for example a normal (Gaussian) distribution, on the histogram, based on the mean and variance of the histogram peaks. This approach is illustrated in Figure 3-11. By arbitrarily shifting the peak of the Gaussian distribution to correspond to the highest bar of the histogram it was, in this case, possible to produce a fit reminiscent of the curves sometimes fitted to histograms by eye (see Figure 3-5).

PREFERENCE CURVES

Data collected as observations of the physical or chemical environment at the location an organism is observed, may indicate preference for the particular set of conditions or may simply be reflective of the existence of those conditions where the organism would be anyway. Since all of the suitability index curves discussed above are predicated on the thesis that organisms seek out preferred conditions and do better under them, some authors have attempted to test their curves to satisfy

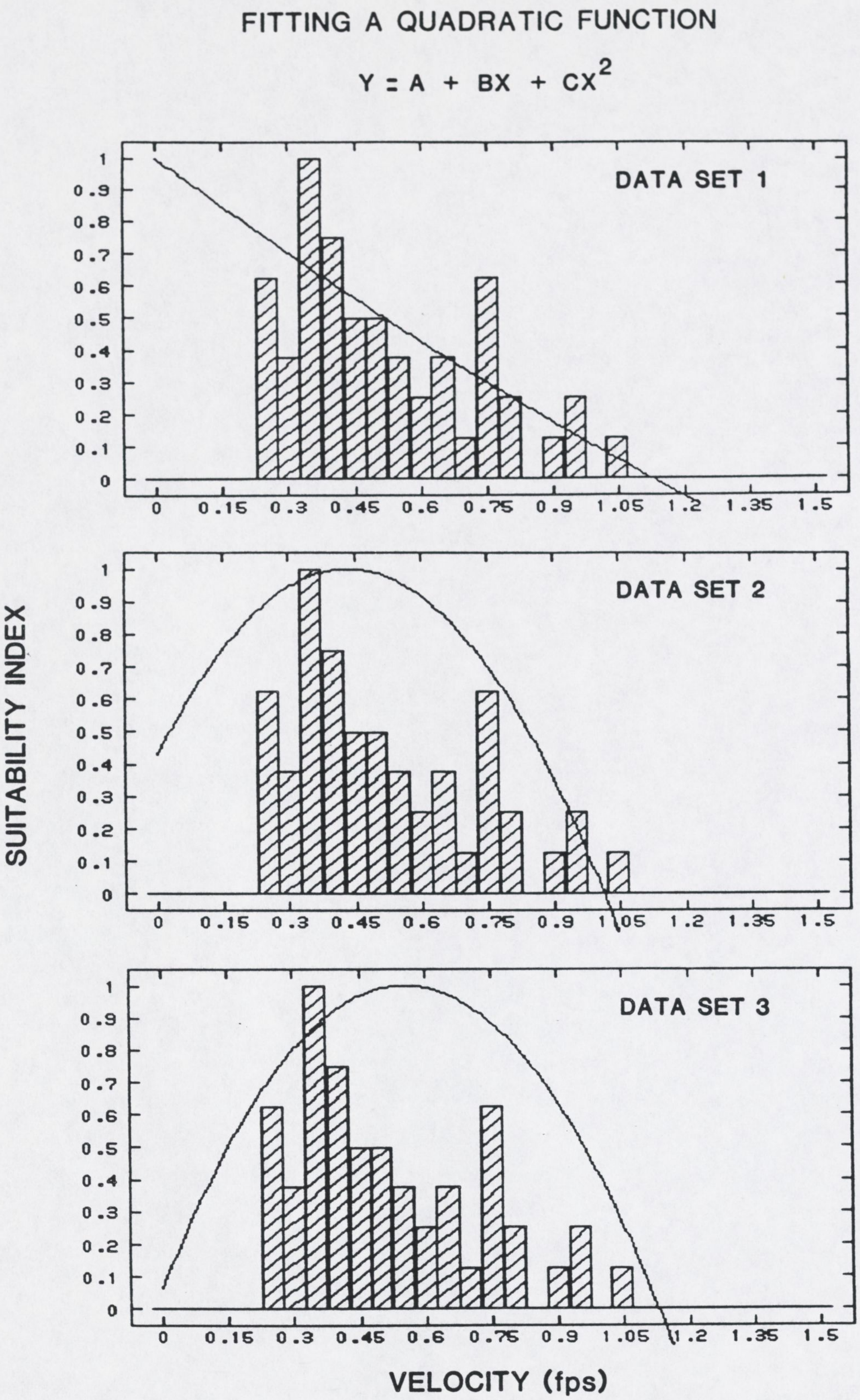


Figure 3-9. An example of the results of fitting a second order polynomial equation to histogram peaks, then normalizing it to produce a biological transformation (Suitability Index) curve. In Data Set 1, no zero suitability values were included outside the range of nonzero values, and the curve did not decline as velocity neared 0. By including zero suitability at zero velocity (Data Set 2) and subsequently adding leading and trailing zeros (Data Set 3), the curve was forced toward zero suitability at zero velocity. (From Hanson, Morhardt, and Coulston 1985).

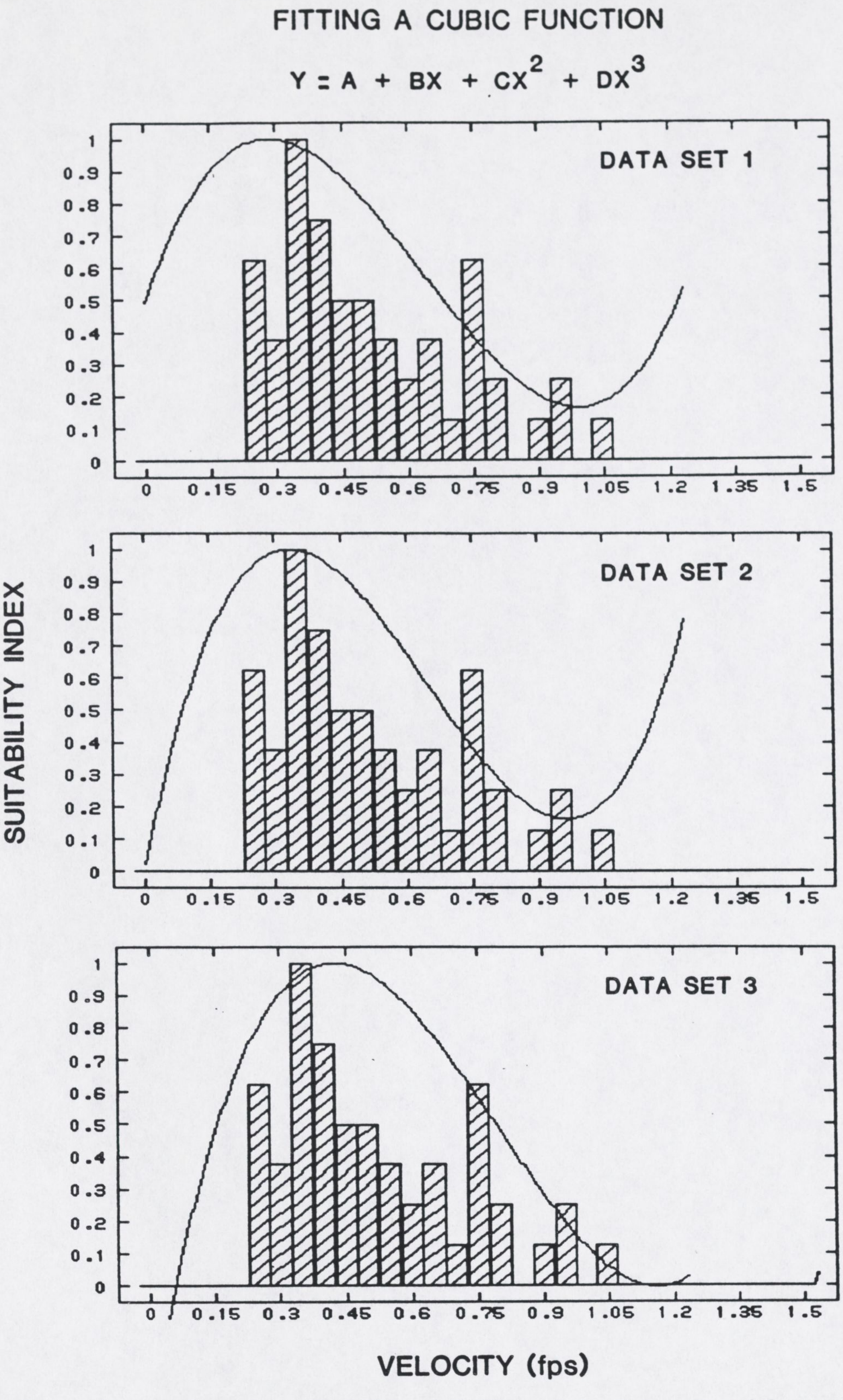


Figure 3-10. An example of the results of fitting a 3rd order polynomial equation to histogram peaks, then normalizing it to produce a biological transformation curve (suitability index). Data Set 1 (top curve) had no leading 0 values in front of the first bar, and the left hand side of the curve reached high values at lower velocities than shown by the histogram. By constraining the curve to pass through 0 (Data Set 2) or by adding leading and trailing 0's (Data Set 3) a better fit was achieved. Note tht the cubic function behaves unrealistically as velocities increase, and should not be used to extrapolate past the data. (From Hanson, Morhardt, and Coulston 1985).

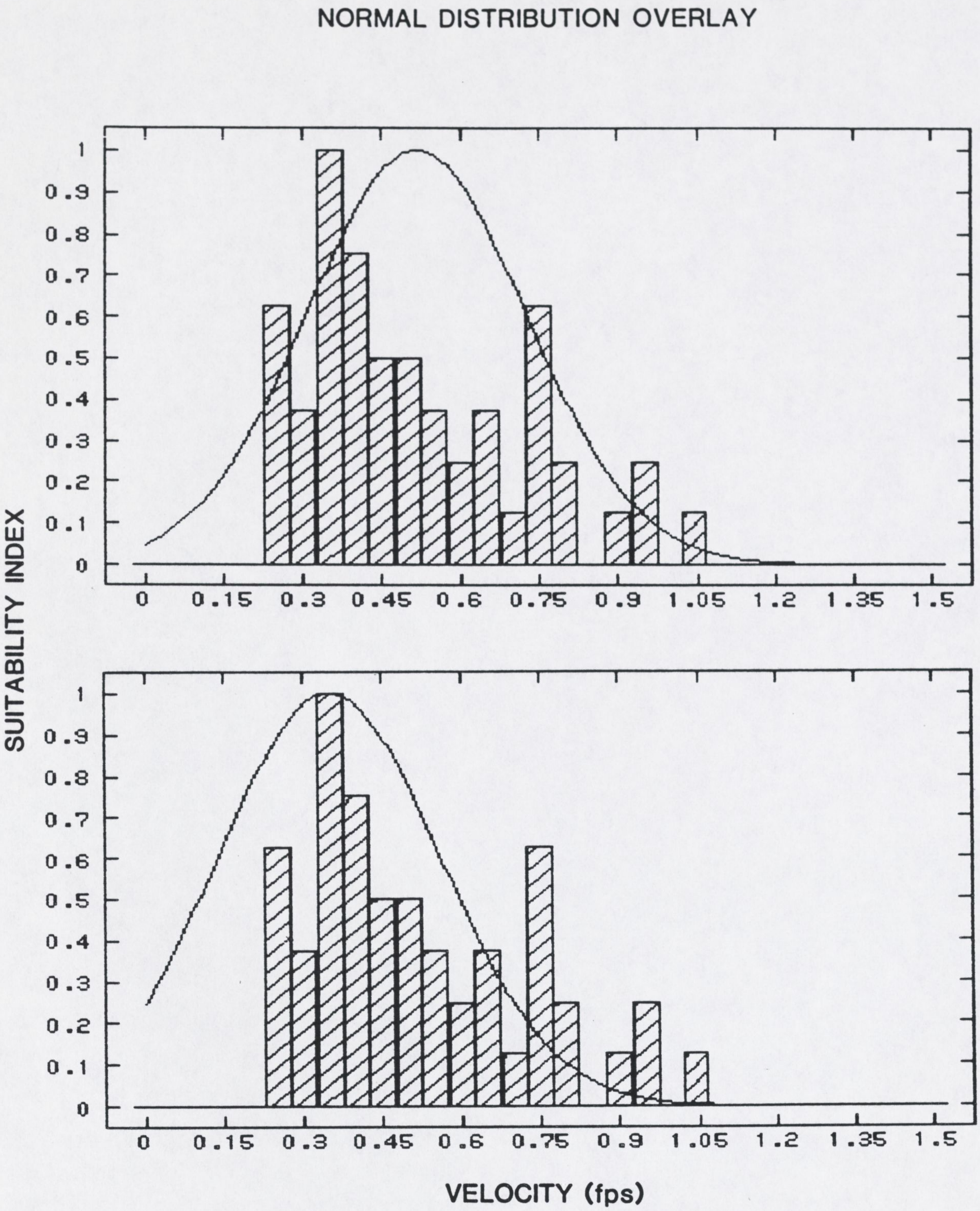


Figure 3-11. Example of a normalized Gaussian distribution overlayed on histogram data to produce a biological transformation curve. In the top graph the distribution clearly does not reflect the distribution of the peaks of the bars. By shifting the curve so that the peak corresponds to the highest bar, the line strongly resembles some of the SI curves fit by eye and appearing in the literature.

themselves that the curves really do reflect preference, and others have devised transformation techniques intended to delete others have devised transformation techniques intended to information not reflective of preference from the curves.

Shirvell and Dungey (1983) satisfied themselves that their observations of depth and velocity at the locations of brown trout did reflect preference because velocity and depth data taken at random stream locations were distributed differently than the depths and velocities where the fish were. As an example, compare the histograms of velocity across transects in a stream (Figure 6-10) with the velocities where fish are located (Figure 3-11). The distribution of velocities in the stream changes with discharge and becomes more evenly distributed as discharges increase. Fish (at least the rainbow trout whose velocity distribution is shown in Figure 3-11) tend to occupy a subset of the available velocities and consequently, can be judged to be selective.

Is there any need for further manipulation of the data to convert the utilization histogram to a preference histogram? Some authors have suggested dividing the utilization data for a variable by the availability of the variable in the stream as a means to eliminate any bias. (Baldridge and Amos 1981, Voos et al. undated). Figures 3-12 and 3-13 are examples of how this affects the utilization curve. In Figure 3-12, normalized utilization data for velocity were divided by normalized velocity distribution data (from Figure 6-10). Note that when the distribution of available velocities is strongly skewed toward the left in the stream, as was the case at 5 cfs, the division procedure shifted the preference curve to the right of the utilization curve. But under the more even available velocity distribution occurring at 50 cfs, the utilization and preference curves had similar shapes. Figure 3-13 shows the results of a similar procedure on the same data. Three-point normalized running means were fitted to both the availability and utilization data prior to making the division. Notice that in this case also, the preference curve based on 5 cfs velocity availability data is much different than that at 50 cfs. Note that in these examples when utilization data had values of zero, we caused preference to have a value of zero as well, otherwise the division would have caused the preference curve to go to infinity. This potential result can create extremely erratic behavior, particularly at the tail end of the curves where there may be little data. As a result, preference may artifactually appear highest where there is little habitat and little utilization.

But must know flow that util. data were collected!

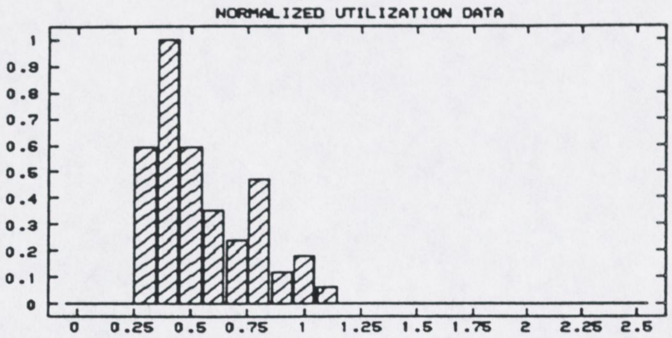
No, when avail. is zero!

But this may be true biologically, i.e. dom. fish use deepest water.

It is by no means clear that the shift produced in the utilization function by division by the 5 cfs velocity data is an accurate reflection of the true preference but it is quite clear that the discharge at which the distribution of velocities in the stream is measured have a profound effect on the shape of the

must do this at same flows util. data measured!

Figure 3-12. An example of the effect of dividing utilization data by availability data in an attempt to correct for insufficient availability of pre-ferred habitat. The resulting preference curves are strongly dependent on the discharge at which the availability data were taken.



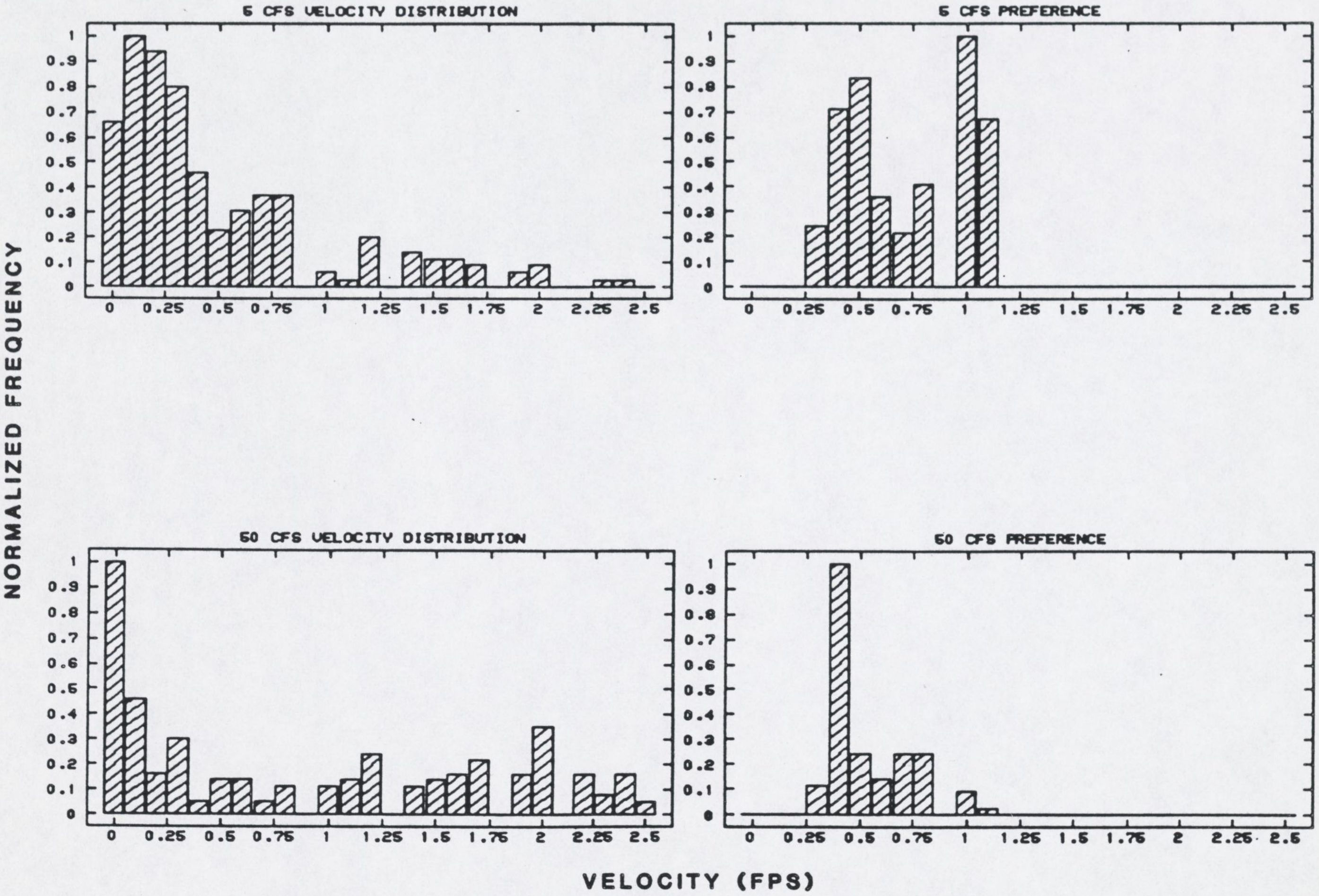
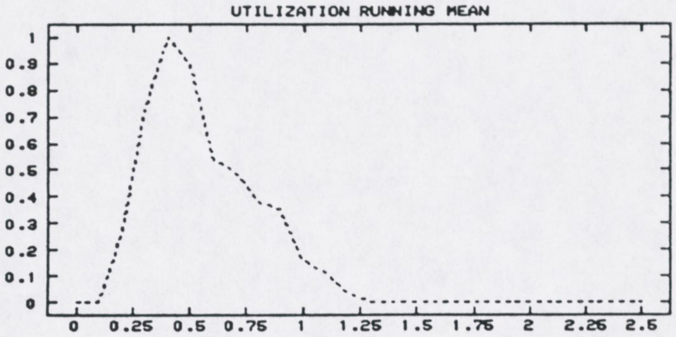
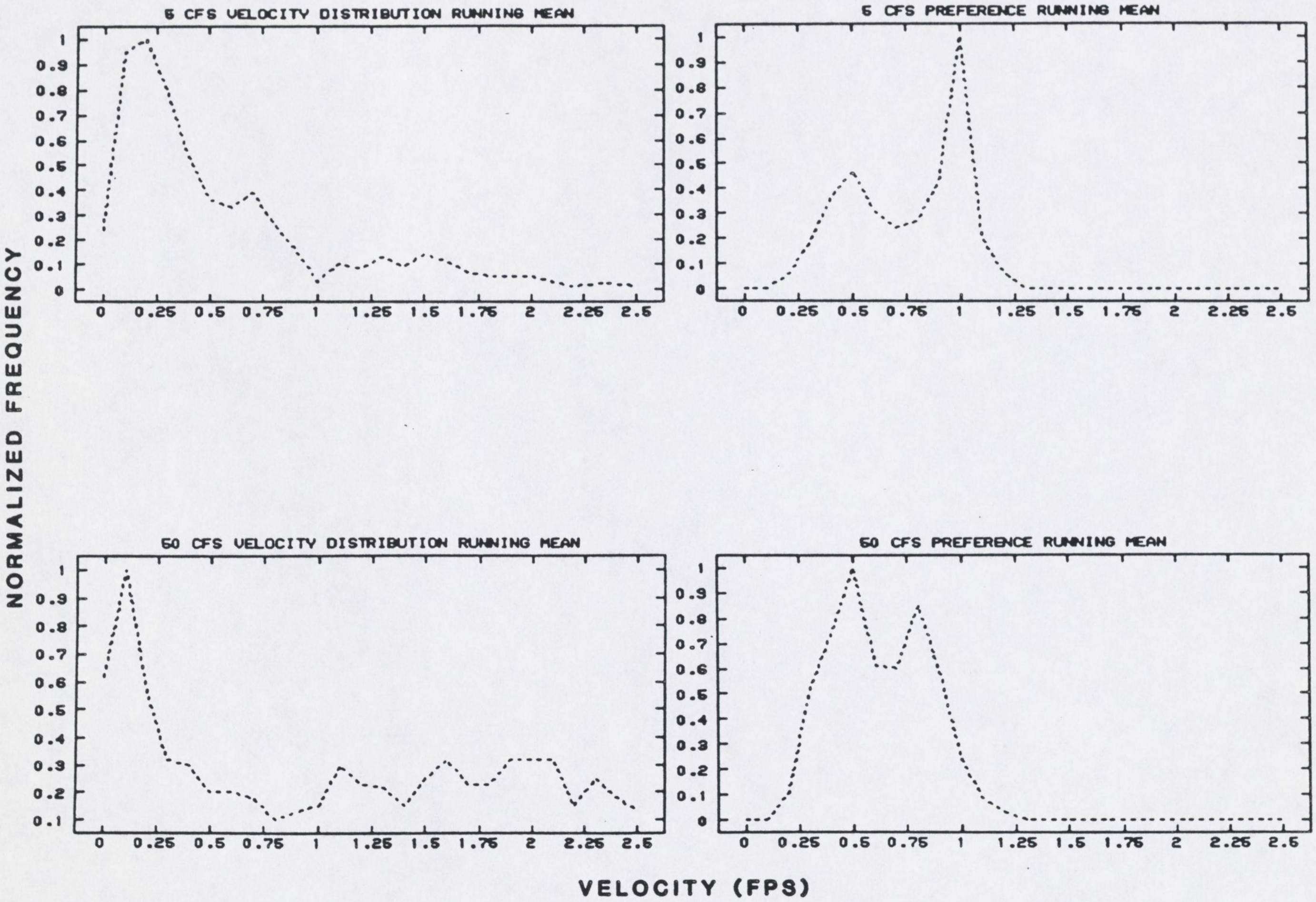


Figure 3-13. An example of the effect of dividing utilization data by availability data in an attempt to correct for insufficient availability of preferred habitat. These curves are based on the histograms of Figure 3-13, but were fit with curves using a normalized 3-point running mean procedure prior to division. Note that the shape of the preference curve is dependent on the discharge at which the availability data were measured.





preference curve. Although this procedure makes some intuitive sense, as far as we know there is no published data indicating that it has any biological significance and in view of the erratic results it can produce, it would seem that it should be used only with extreme caution.

Another problem with this approach is the difficulty one encounters in making a decision on where to measure the distribution of velocities and depths in a stream. The problem is most obvious in the case of spawning salmonids. We have observed rainbow trout, for example, to be highly selective in their choice of spawning locations, spawning only in a very short segment of stream and ignoring miles of habitat much of which has apparently identical depth, velocity, and substrate characteristics. The fish are clearly being selective, but division of the utilization data by velocity distribution data throughout miles of unused stream would be exceedingly misleading, since some other factor has caused the selection.

Kelley et al. (1985) noted a similar situation with regard to selection of spawning sites by chinook salmon. Although suitable substrates, depths, and velocities occurred over much of the river he was examining, the salmon only spawned in the riffes and ignored similar habitat occurring in runs.

Another technique for correcting utilization data to reflect preference has been suggested in a manuscript distributed by the USFWS Instream Flow Service Group (Nelson 1984). This document suggests fitting polynomial equations to both utilization data and to data on the distribution of velocity and depth in the stream, then subtracting the polynomial regression coefficients of the availability data from those of the utilization data. The intended effect of this procedure is illustrated in Nelson (1984) with plots which, however, do not reflect at all the actual behavior of the equations they are supposed to be illustrating. Nelson offers no justification for this procedure, and we have been unable to see any merit in it.

BIVARIATE SUITABILITY INDICES

There has been some work toward producing bivariate suitability indices using just depth and velocity, for use in the FWS HABTAT model (Voos 1981, Voos et al. undated). This work has been done in response to the kind of data shown in Figures 6-7 and 6-8, from which it appears that fish do not select depth and velocity independently of one another, but select suitable combinations of depth and velocity. These bivariate suitability functions are a logical first step toward the multivariate models which are likely to be necessary to predict population dynamics, but so far they have not received much attention, or been shown to produce any more realistic results than the univariate models.

EMPIRICAL MODIFICATION OF SI CURVES

A potentially valuable technique, which as far as we are aware has received no attention at all, is the empirical adjustment of the shape of empirically derived SI curves. It is entirely possible that models which use species-specific and location-specific biological transformation curves could be made more realistic by changing the shapes of the curves. Most of the work that has gone into development of empirical depth and velocity preference curves has been directed toward assuring that the curves reflect the preferences of the resident fish (for example Aceituno et al. 1985) or to increasing the precision with which the actual velocities at the nose of the fish are recorded (Studley, unpublished). It may be, however, that although depth and velocity affect fish populations, existance of the preferred depths and velocities do not. Adjustment of the preference curves specifically to increase predictability of standing crop would be a legitimate activity, comparable to estimating parameters in least squares regression models.

OTHER WAYS TO LINEARIZE DATA

When data linearization does not have much effect on the scatter of data, other approaches have been tried, sometimes with good success. Binns and Eisermann (1979) transformed all their input data, but as can be seen from a comparison of the untransformed and transformed values of five of their input variables (Figure 3-14) there is little increase in linearity. They then produced new variables by systematically multiplying the various transformed variables together, and, suprisingly to us, discovered a combination of four decidely scattered input variables which, when multiplied together, produced a remarkably linear new variable (Figure 3-15). There are a wide variety of strategies of this type available (see Draper and Smith 1981 or any other textbook on regression analyses for examples). Instream flow modeling may benefit from applying some of them.

But do they make any sense biologically?

SUMMARY

The object of transforming data to biological suitability curves is to linearize the data as much as possible and this is accomplished by making as good a fit to the data as possible, regardless of the technique used. This can be done both by using mathematical curve fitting techniques or fitting the data by eye. Using some form of mathematical technique is preferable for two reasons. First, it results in a curve which is already mathematically described at the time it is fit, and does not require subsequent digitizing to describe and use it. Second, and more important, it removes investigor bias which is often obvious in curves fit by eye, and results in a reproducible curve. The utility of such curves, though is less clear. It is possible that modifying the shape of the curves to increase predictability would advance the usefulness of such models.

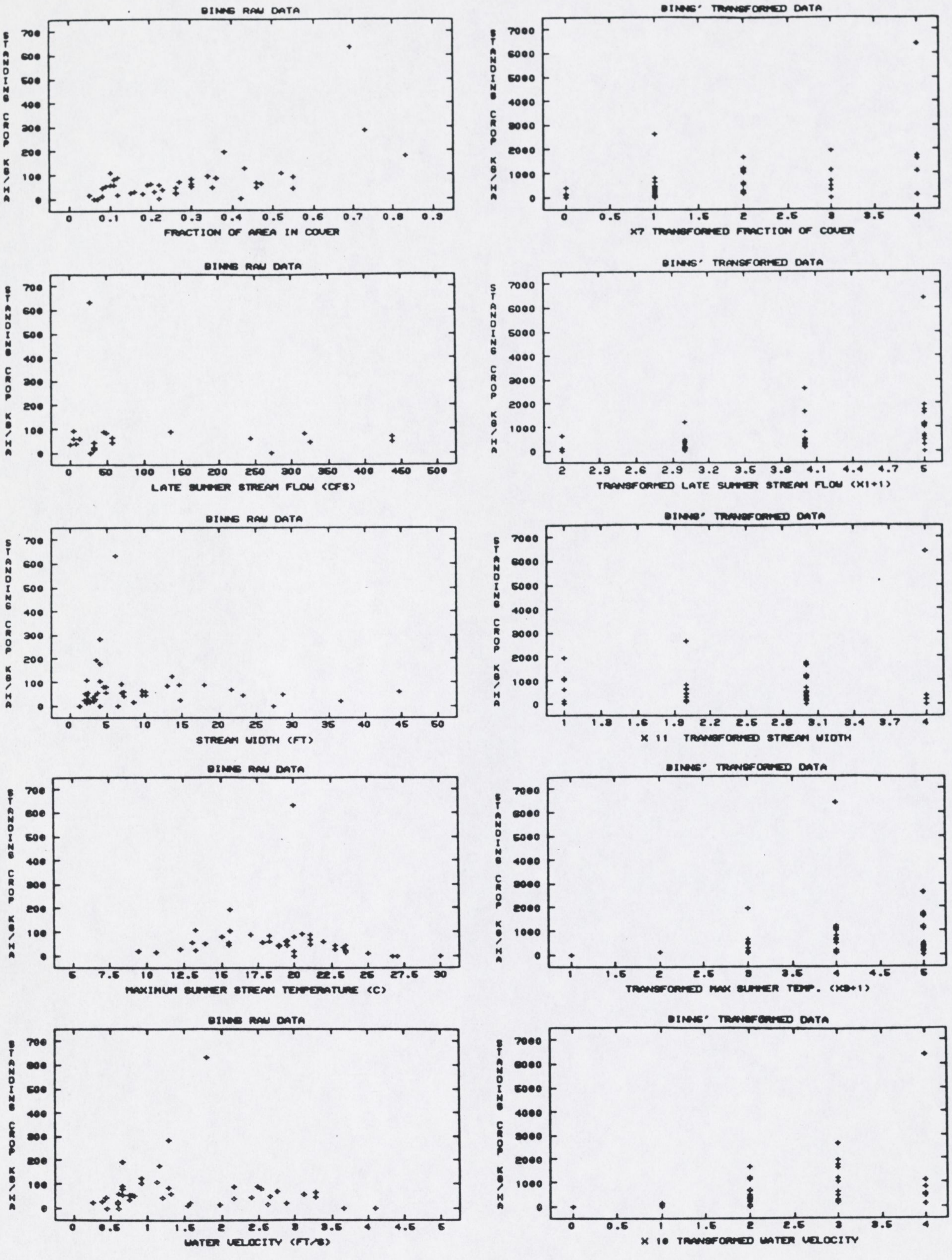


Figure 3-14. The effects of Binns' transformation of his raw data into slightly more linear form prior to model construction.

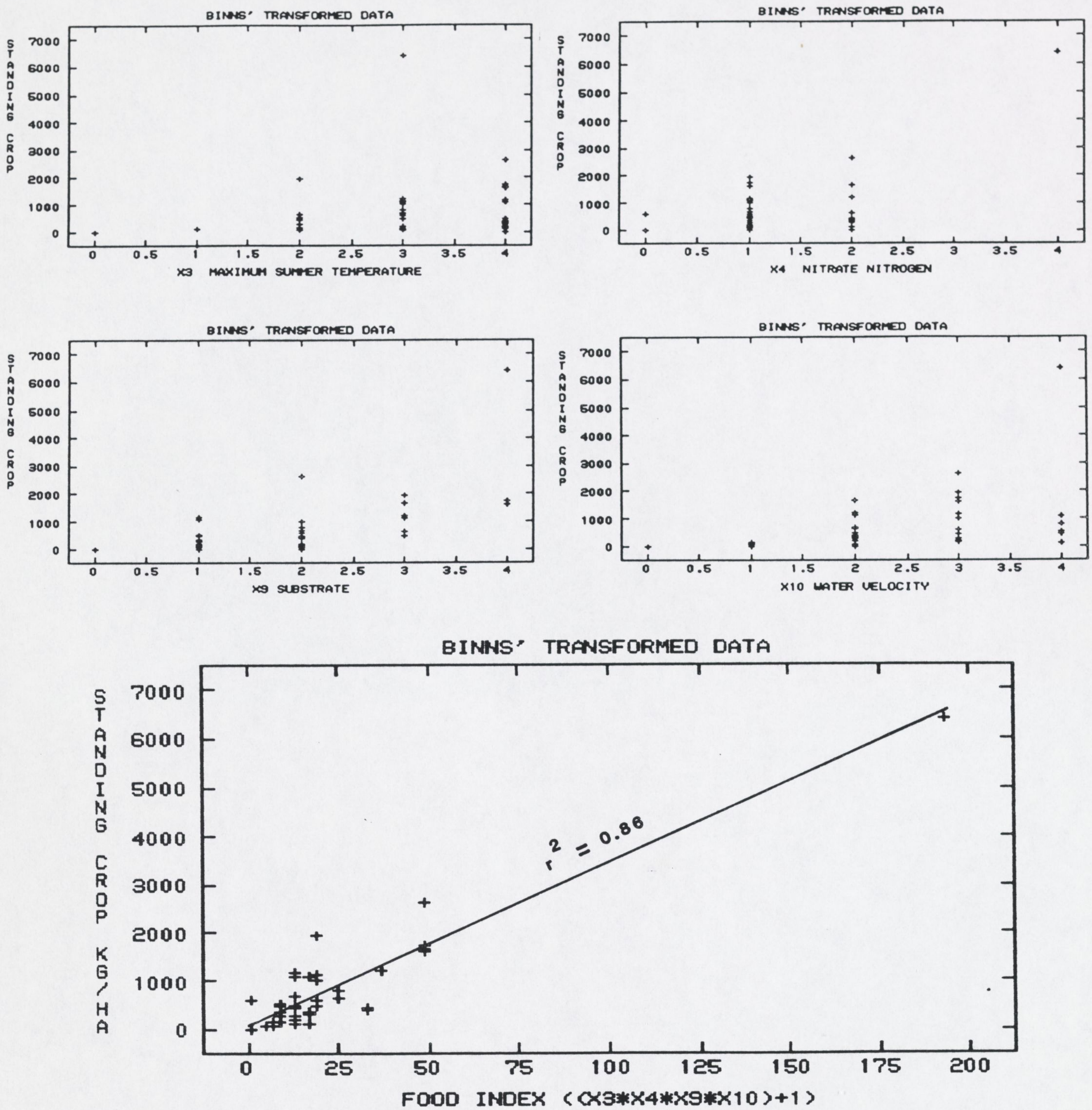


Figure 3-15. An example of the generation of a new variable more strongly correlated with the response variable by multiplying input variables together. Binns and Eiserman (1979) took 4 transformed variables not strongly correlated with standing crop (the 4 small graphs) and produced a new variable (F) strongly correlated with standing crop (kg/km) shown in the large graph. Source of the data in these plots in Binns 1979.

CHAPTER 4: CHOOSING THE MODEL STRUCTURE AND ESTIMATING PARAMETERS

The majority of the models and methodolgies reviewed in this report can be expressed in the form of mathematical equations and the structure of these equations varies from model to model.

REGRESSION MODELS:

Most of the standing crop models and a number of the habitat quality models are linear regressions of the form:

$$Y = p_o + p_1V_1 + p_2V_2 + p_3V_3 \ldots p_nV_n$$

where Y is the response variable

p_n is the parameter (regression coefficient) associated with each input variable, and

V_n is the value of the input variables.

Models of this type are almost always formulated by collecting data on both the response variable and the input variables and then estimating the values of the parameters using linear least squares regression methods. The term "parameter" as used in this report refers specifically to these numbers which become a permanent part of the model and which are estimated from the values of input and response variables in the data set used to formulate the model. They could equally well be called coefficients, and the information they contain is the relative importance of a change in each of the input variables to the ultimate value of the response variables. They can be negative as well as positive and if so, indicate that as the input variable increases in value, the response variable decreases.

ESTIMATING PARAMETERS

The purpose of least squares linear regression or multiple linear regression procedures is to estimate the most suitable values of these parameters from the data available. For multiple linear regression, which is necessary if more than one input variable is to be used, the values of the parameters influence one another, and if any of the variables is deleted and the least squares procedure redone, the values of the parameters for the remaining variables would change.

There is usually a hierarchy of importance that can be assigned to the input variables with some contributing strongly to the explanation of the variability of the response variable and some having little effect. Most modelers prefer to omit variables that have little effect on the descriptive ability of the model. Various statistical criteria are available to serve as guidelines as to whether to include a variable. Binns and Eiserman (1979), for example included only variables which had a multiple correlation coefficient (R) of 0.28 or greater. Rabern used

Mallow's "CP" statistic; and it is common to set a value of the F statistic (connoting significance), or the percent change in the multiple correlation coefficient with an additional variable, as criteria for including variables. Adding variables never decreases the value of R, but the amount of improvement may not warrant the effort required to use them. Sometimes modelers use combinations of transformed and untransformed variables as well, all with the intent of explaining as much of the variation as possible.

Another technique frequently used is to take logarithms of some or all of the terms resulting in models that are the equivalent of exponential or power functions. Binn's and Eiserman's Model II, for example is a power function but was arrived at by using multiple linear regression to estimate the parameters on log-transformed input and response variables.

EFFECT OF TRANSFORMATION ON MODEL OUTPUT.

As an example of the effects of some of these activities in improving model performance, we have used the same data set used by Binns and Eiserman (1979 - see Figures 3-14 and 3-15) to develop comparable models from raw data, biologically transformed data, and biologically transformed data using new variables made by multiplying old variables together.

Figure 4-1 is a scatter diagram of an index (which we called EA's HQI) derived by subjecting all of Binns and Eiserman's raw data to interactive stepwise multiple linear regression. The inclusion of just two terms, nitrate nitrogen and percent cover, explained 66 percent of the variability, and the additional terms did not increase the value of R^2 by more than one percent each, and so were not included. * Note that the regression was strongly influenced by a single high value. It is potentially misleading to include values that are quite different from the rest of the data set, and all of the analyses using this data set, including Binns and Eisermann's (1979) Model II, appear stronger than they should because of this single high value. The value may be correct, but the presence of only one variable in the upper and right halves of the regression plots gives this single value a disproportionate importance.

By using Binns and Eiserman's transformed (but not grouped) data (see Figures 3-14 and 3-15), a multiple linear regression including nitrate nitrogen, percent cover, number of benthic organisms per square meter and maximum annual flow minus minimum annual flow (annual flow variation) produced an estimated standing crop (termed a Habitat Quality Index in Binns and Eiserman's parlance) correlated with the actual one with an r^2 of 0.78, explaining 78% of the variation in standing crop, an improvement over the data which were not biologically transformed (Figure 4-2). The single variable most strongly correlated with

Transformed means rating variables.

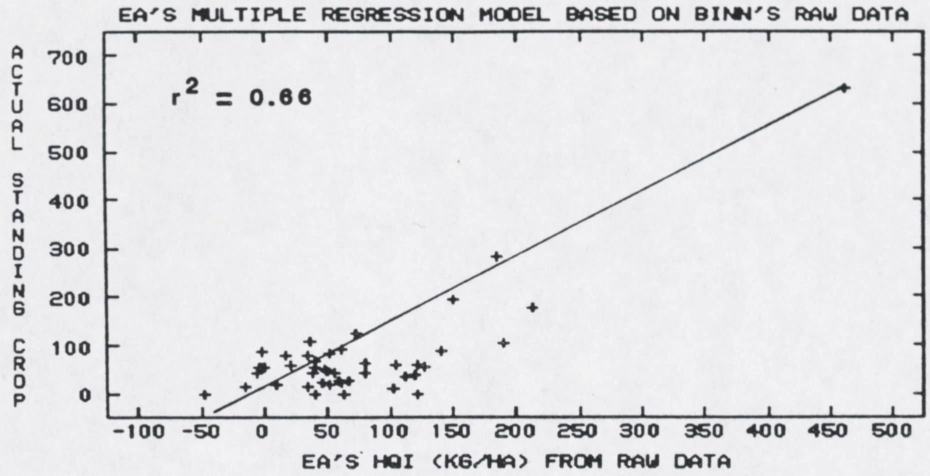


Figure 4-1. The results of the best two variable (% cover, nitrate nitrogen) multiple linear regression model using some of Binns' (1979) untransformed data. Note that although the model explains 66% of the variability of standing crop, it appears to explain almost none of it below approximately 100 kg/ha. The addition of % cover and velocity terms had < 1% improvement in the R^2.

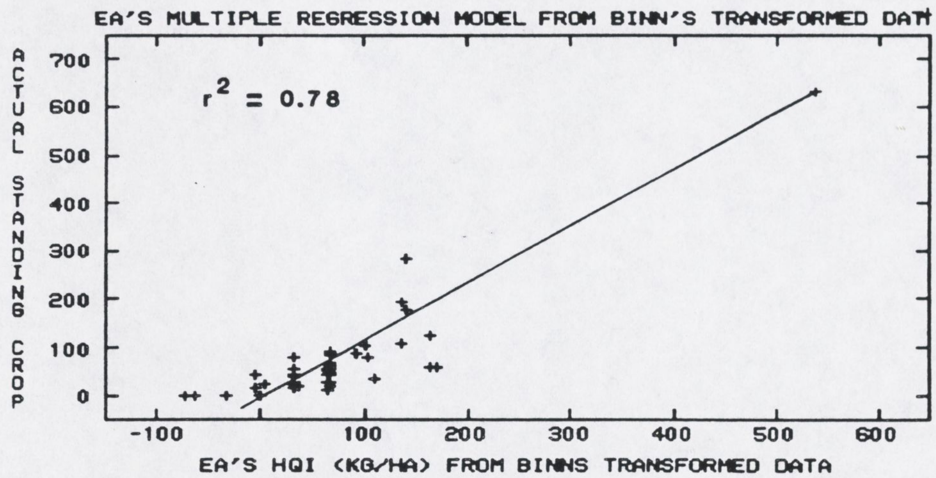


Figure 4-2. The results of the best three variable (annual flow variation, nitrate nitrogen, # organisms/m^2) multiple linear regression using Binns' (1979) transformed data. Additional terms made almost no difference in the R^2.

standing crop was the annual flow variation term which explained 40% of the variability of standing crop.

We then used the same biologically transformed and multiplied variables Binns and Eiserman used in their Model II (but without log-transforming them into a power function) and achieved an r^2 of 0.93 indicating very little scatter around the regression line, and a model that explained 93 percent of the variation in standing crop (Figure 4-3). Not surprisingly, the F variable (described in Figure 3-15) explained 88% of the variability by itself, with late summer flow, annual flow variation and late summer temperature having almost no effect at all. This is quite different from the effects using the transformed or multipled variables in which annual flow variation had the greatest effect.

Finally, for completeness, we log-transformed both input and response variables and again ran a multiple linear regression to duplicate Binns and Eisermann's Model II. In this case, the log transformation decreased the correlation of standing crop with the F variable to and r^2 of 0.80, but ultimately resulted in an r^2 of 0.97. It is important to note that although Binns and Eisermann refer to the x axis of this regression as "Predicted Standing Crop", it is not, and this regression does not constitute a test or validation of the model. Instead the r^2 value is merely a description of how well this model fits the data from the 36 sites that were used to create it (Figure 4-4).

There are many manipulations that can be done with input data and model structure to attempt to improve the discriptiveness of the model. Binns and Eisermann's 1979 paper is the only habitat quality or instream flow model that has explored these enhancements, and other models might be significantly improved by following suit.

CONCEPTUAL MODEL STRUCTURE AND PARAMETER ESTIMATION

The structure of many habitat quality models is neither empirical in the regression modeling sense nor mechanistic in the sense that it reflects the way the modellers thought the terms interacted. Therefore we have called these structures conceptual rather than mechanistic.

Some of them are fairly simple and involve only multiplying the transformed variables together much like Binns and Eiserman (1979) did for their F variable. The most prominant example is the FWS HABITAT Model.

Others are more complex, for example Wesche's WRR1 Trout Cover rating and Orsborn's Maximum Spawning Area Method. The most complex are the FWS HSI models (identified in Table 2-1). These use a variety of seemingly arbitrary combinations of arithmetic means, and geometric means of as many as 20 input variables.

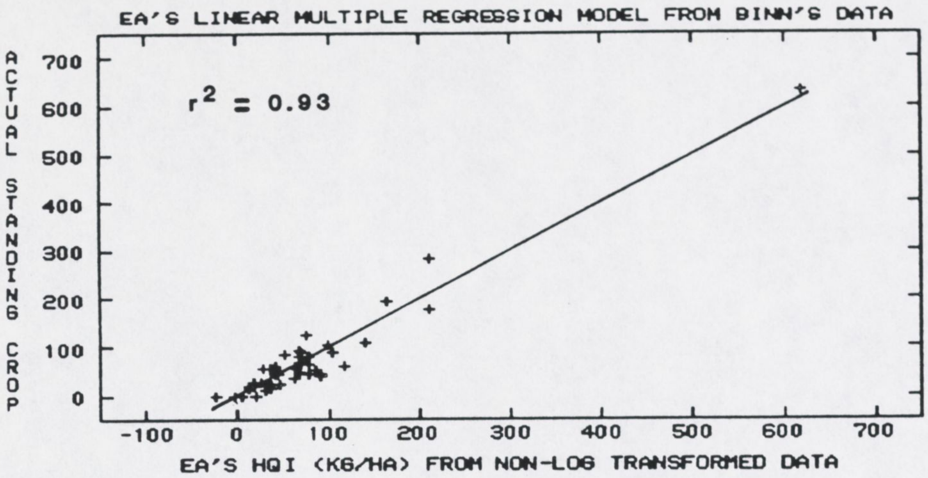


Figure 4-3. The results of multiple linear regression using the same transformed variable used by Binns and Eiserman (1970) in their Model II, but not log-transforming them before applying the multiple linear regression model. Note that the results are quite similar to the log transformed model settled on by Binns and Eiserman (below).

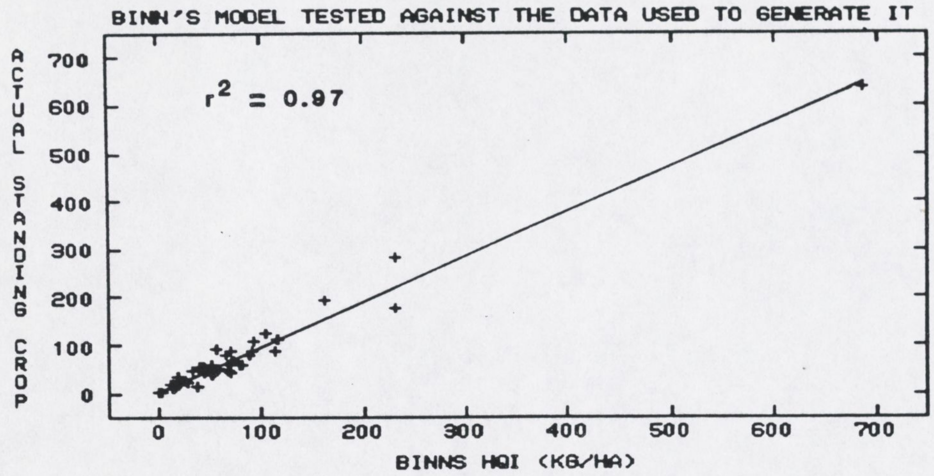


Figure 4-4. The results of Binns and Eiserman's (1979) Model II showing that there is little scatter around a regression between modeled and actual standing crop. Note that this type of graph does not constitute a test of the predictivness of the model, since the data on the horizontal axis have been generated partially from the data on the vertical axis. It does, however, show that the model explains a large part of the variability and is therefore, highly descriptive.

*

For none of these conceptual models is there any discussion or rationale for the particular structure used, and since the response variable is unmeasureable there is no direct way to test the performance of the model.

It is also notable that these models generally are without parameters (i.e. all variables have their parameters set arbitrarily at 1.0) indicating the modellers assumption that all variables have equal value. In one **regression** model the authors (Nickelson et al. 1979) even changed all parameters to 1.0 because they said they had no biological meaning and did not result in a unique solution to the equation. Since regression modeling makes it abundantly clear that this is almost never the case, it is surprising that there are not more attempts to assign parameters conceptually.

CHAPTER 5: DETERMINATION OF MODEL SENSITIVITY

An important aspect of instream flow models is their sensitivity to changes in the values of input variables. There are several general observations that can be made about model sensitivity.

SENSITIVITY AS A FUNCTION OF NUMBERS OF TERMS

The fewer variables in a model, the more sensitive the output is to changes in any one variable. Models with few terms, such as the FWS HABTAT model, produce a much greater response to variation in one of their input variables than do models with many terms such as the FWS HSI models. This has its good and bad aspects. If one is only interested in the effects of one or a few variables, then the decrease in model sensitivity caused by a proliferation of terms is undesirable and serves only to interfere with predictiveness. But, if there are a number of input variables contributing significantly to the response variable, utilization of a smaller number is likely to render a model less descriptive and less predictive.

An adverse aspect of using large numbers of terms is that they can interact in non-intuitive ~~and~~ ways that are difficult to explain. This is particularly true for models which are entirely conceptual in origin. As an example, we utilized one of the FWS HSI models (Hickman and Raleigh 1982) as an instream flow model by making up hypothetical functional relationships between discharge and 7 of the 19 input variables that we thought would vary with flow (holding all other variables constant). Figure 5-1 shows the functional relationships. We then ran the model over a range of discharges and obtained the curves shown in Figure 5-2. The model behavior is not explicable from observation of the input curves and the response variables as a function of discharge. This example does show, however, that the HSI models work fine as instream flow models, as long as one agrees with their assumptions.

SENSITIVITY AS A FUNCTION OF LOCATION ON THE BIOLOGICAL TRANSFORMATION CURVE

When biological transformation curves are used, they are usually nonlinear and often change rapidly in suitability with discharge at particular flows. For example, in each of the transformation (SI) curves shown in Figure 3-6, very large changes in response (suitability) occur at depths in the range of 1-2 ft, and are strongly dependent on the particular form of the curve accepted. This is an important phenomenon in many instream flow modeling efforts, because the most rapid changes in suitability curves with discharge usually occur at the low flows that are likely to be in contention, and the specific location of the curve can be a critical part of the analysis. Figure 5-3 is an example of the change in the FWS HABTAT response variable (Weighted Usable Area) as a result of systematically shifting one of the biological

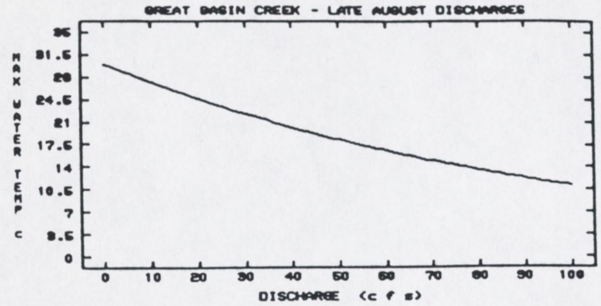


Figure 5-1. Made up environmental variable curves for test using FWS/HSI cutthroat trout model as an instream flow model. (Hickman and Raleigh 1982)

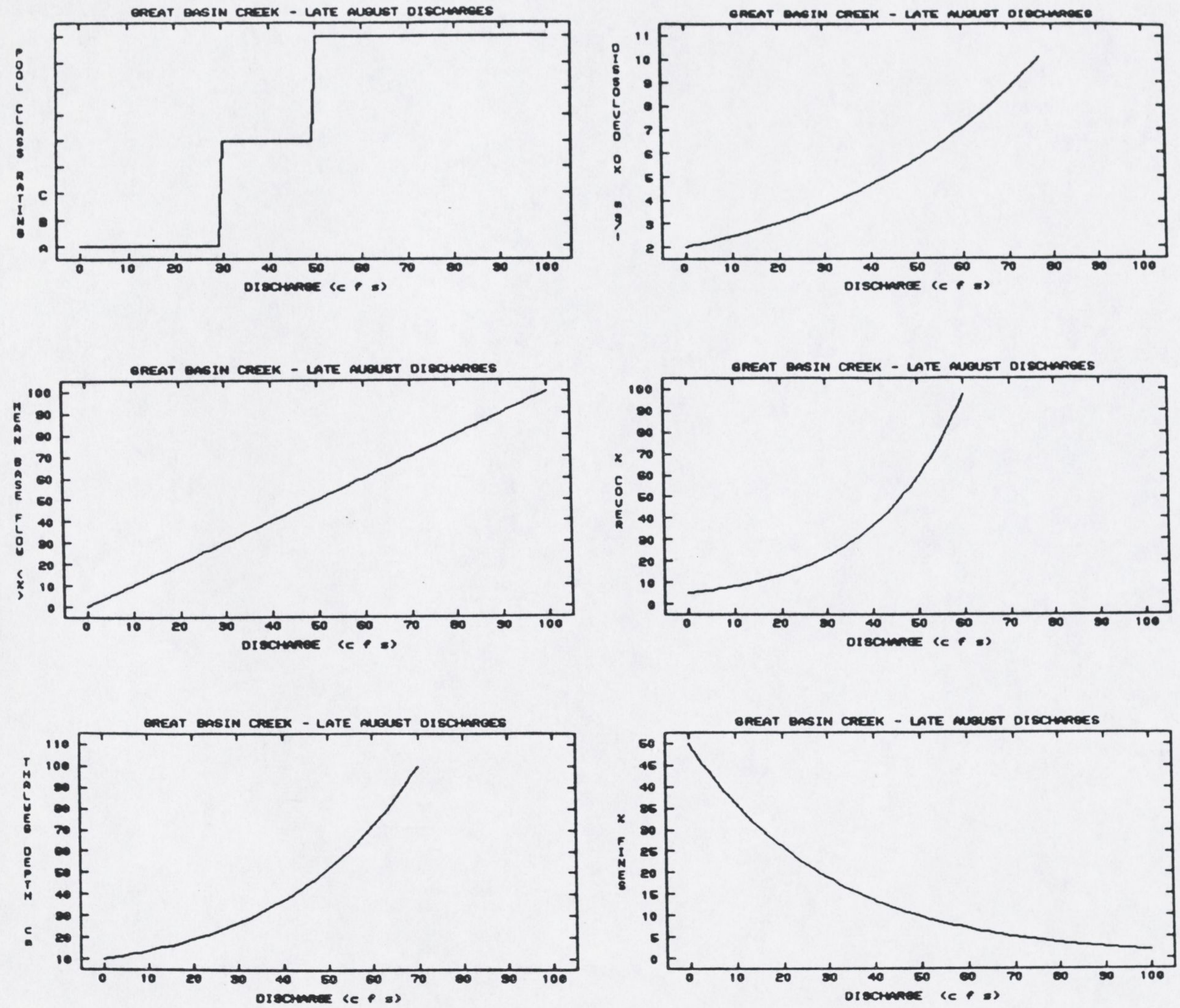
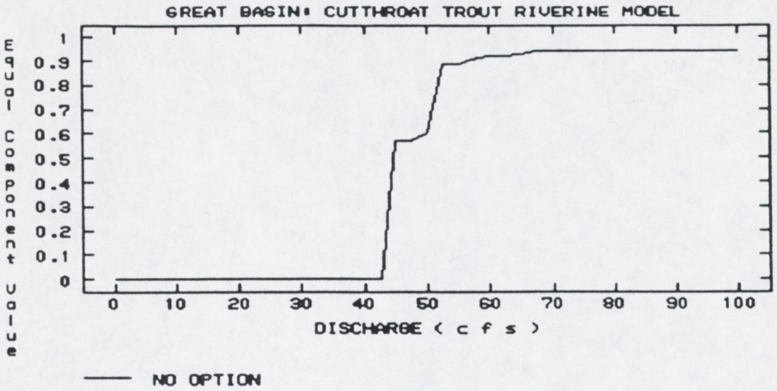
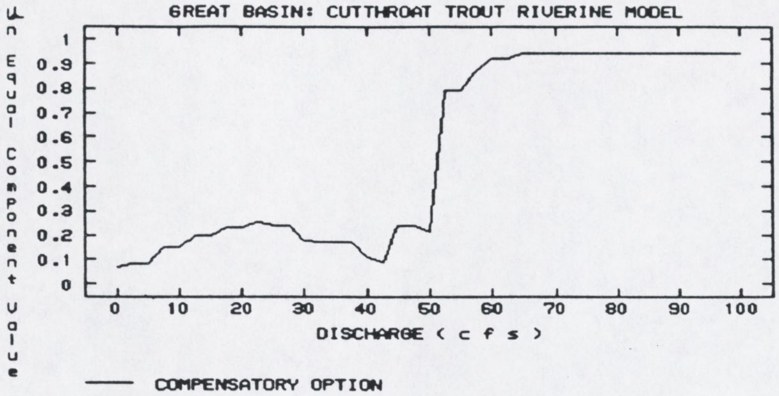
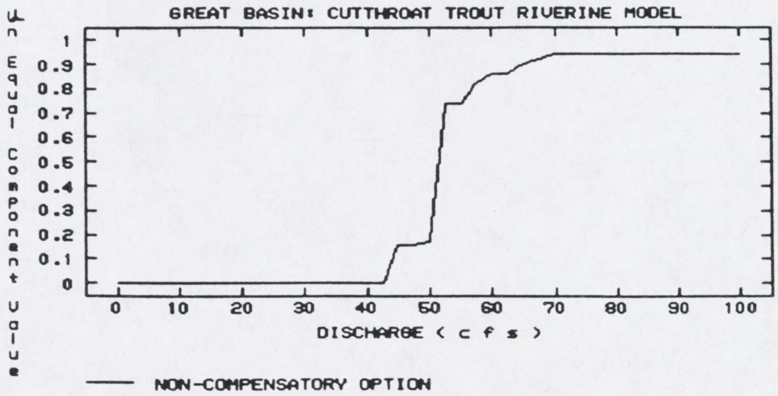


Figure 5-2. The cutthroat trout FWS/HSI used as an instream flow model. (Three internal options are used).







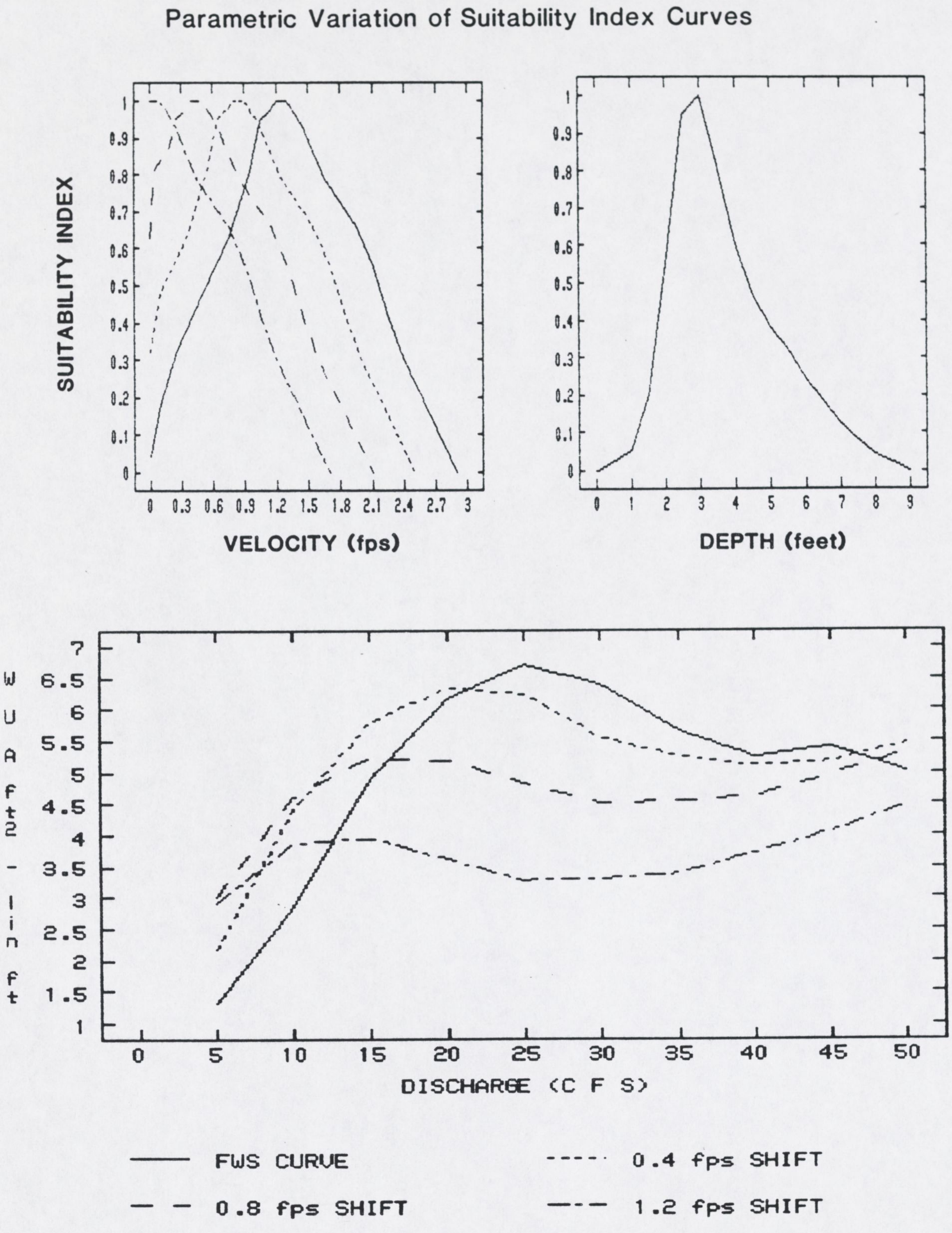


Figure 5-3. An example of the sensitivity of Weighted Usable Area (using the FWS HABTAT model) to shifts in the velocity suitability index. This effect is stream-specific and can be quite pronounced with any SI curve.

transformation curves (velocity) to the left. In this example, the location of the peak of the WUA curve shifted to lower discharges while the absolute magnitude of the WUA index also fell. In some streams the effect would be more pronounced and in others less pronounced, but would always occur and is worth exploring if there is doubt about the appropriate location of the leading edge of one of the SI curves.

A typical situation casting doubt on the leading edge of this type of curve is a preference of smaller fish for shallower, less rapid water. If the fish in the stream in question are smaller than the fish for which the SI curves were generated (as can happen as a result of elevation or productivity), the true SI curve is likely to be shifted to the left, and would thus result in a lower instream flow recommendation.

SENSITIVITY RESULTING FROM INTERACTION BETWEEN INPUT VARIABLES

Figure 5-2 is a good example of an unexpected large sensitivity of the response variable over a small range of discharges. The inverse situation is one in which the changes in one input variable cancel out the effects of the changes of another, resulting in little change in the response variable with changes in discharge. Figures 5-4 and 5-5 show the changes in velocity and depth suitability associated with discharge (resulting from the changes in depth and velocity distributions shown in Figures 6-9 and 6-10 transformed using the original curves in Figure 5-3). These are shown as means with 95% confidence limits as a function of discharge in the top two graphs of Figure 5-6. As discharge goes up, velocity suitability first increases then decreases, but depth suitability continuously increases. When depth and velocity suitabilities are multiplied together for each data point (point on a transect) as functions of discharge as they would be if entered into the FWS HABTAT model, the increases in suitability of depth with discharge counterbalances the decreases in velocity suitability resulting in a fairly stable response variable. A similar response in Weighted Usable Area along with similarly derived confidence intervals, is shown in Figure 5-7 for a different stream.

SENSITIVITY AS A FUNCTION OF THE AMOUNT OF INPUT DATA REQUIRED

Models that use a single datum (such as Tennant's Montana Method) or a few data points such as many of the models using just Basin or Discharge variables, are quite sensitive to variations in the input. Models that use a great deal of input data, such as those based on measurements of variables at many points along transects, are much less susceptable to individual errors, because random errors have an opportunity to balance one another out. In order to test the sensitivity of the IFG4/HABTAT model to random errors in depth and velocity measurements, we conducted a Monte Carlo simulation by introducing normally distributed random variation into individual depth and velocity measurements

Figure 5-4. Frequency distribution of velocity suitability at transect measurement points at discharges from 5 - 50 cfs. The data are from a small Sierran stream and show an interesting increase followed by a decrease in the number of highly suitable areas as discharge is increased.

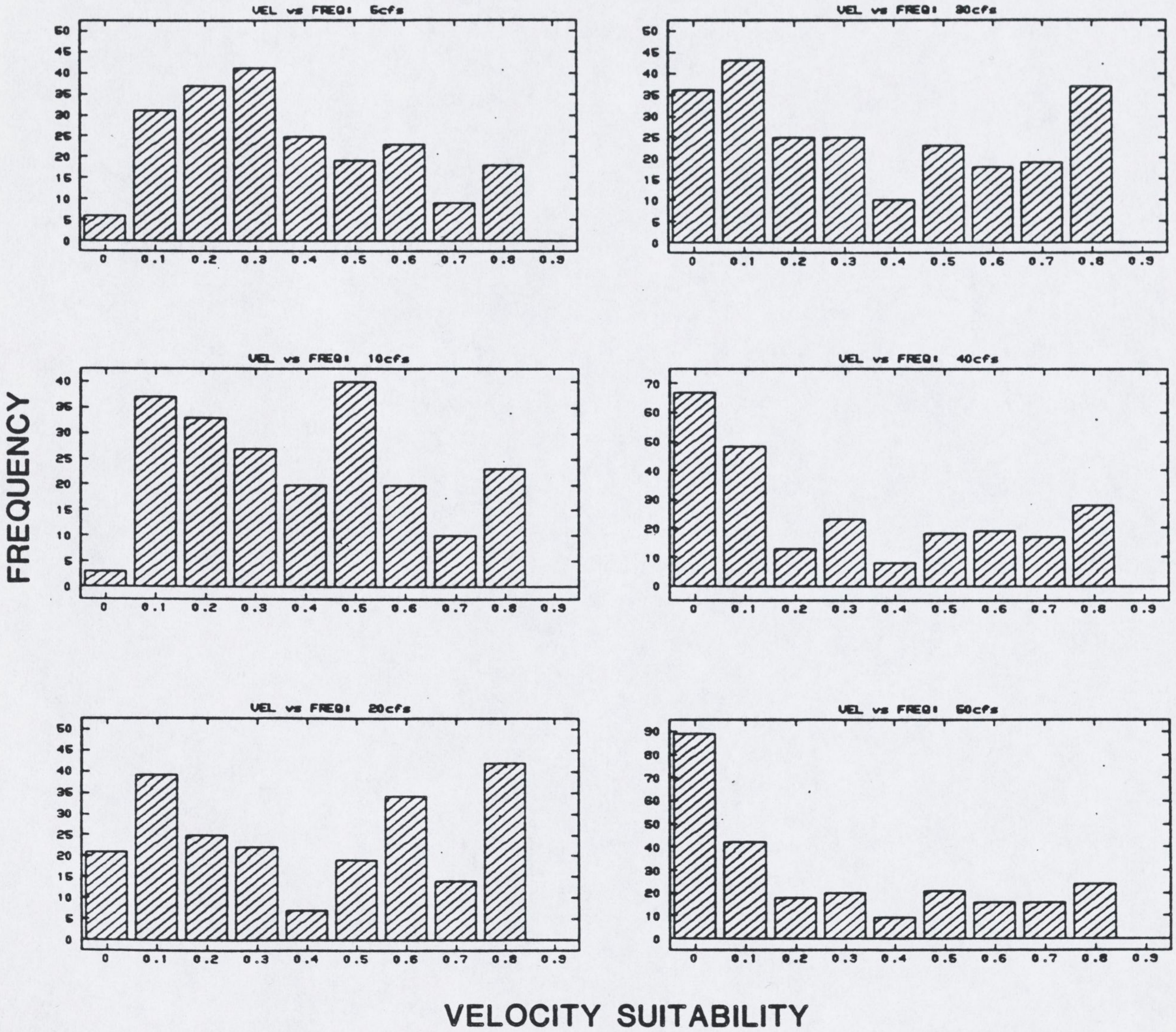
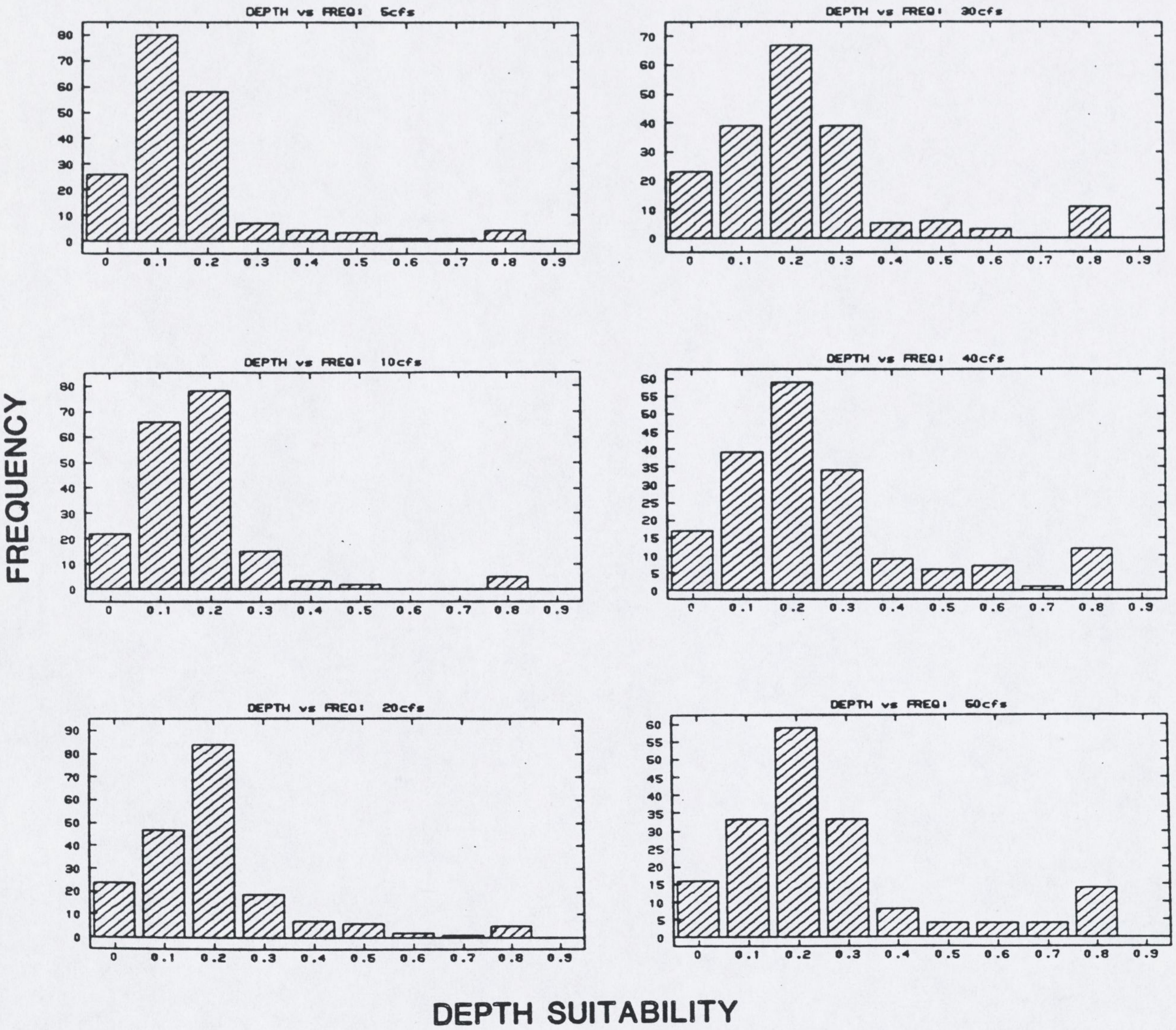


Figure 5-5. Frequency distribution of depth suitability at transect measurement points at discharges from 5 to 50 cfs. The data are from a small Sierran stream.



Typical Distribution of Depth and Velocity Suitabilities

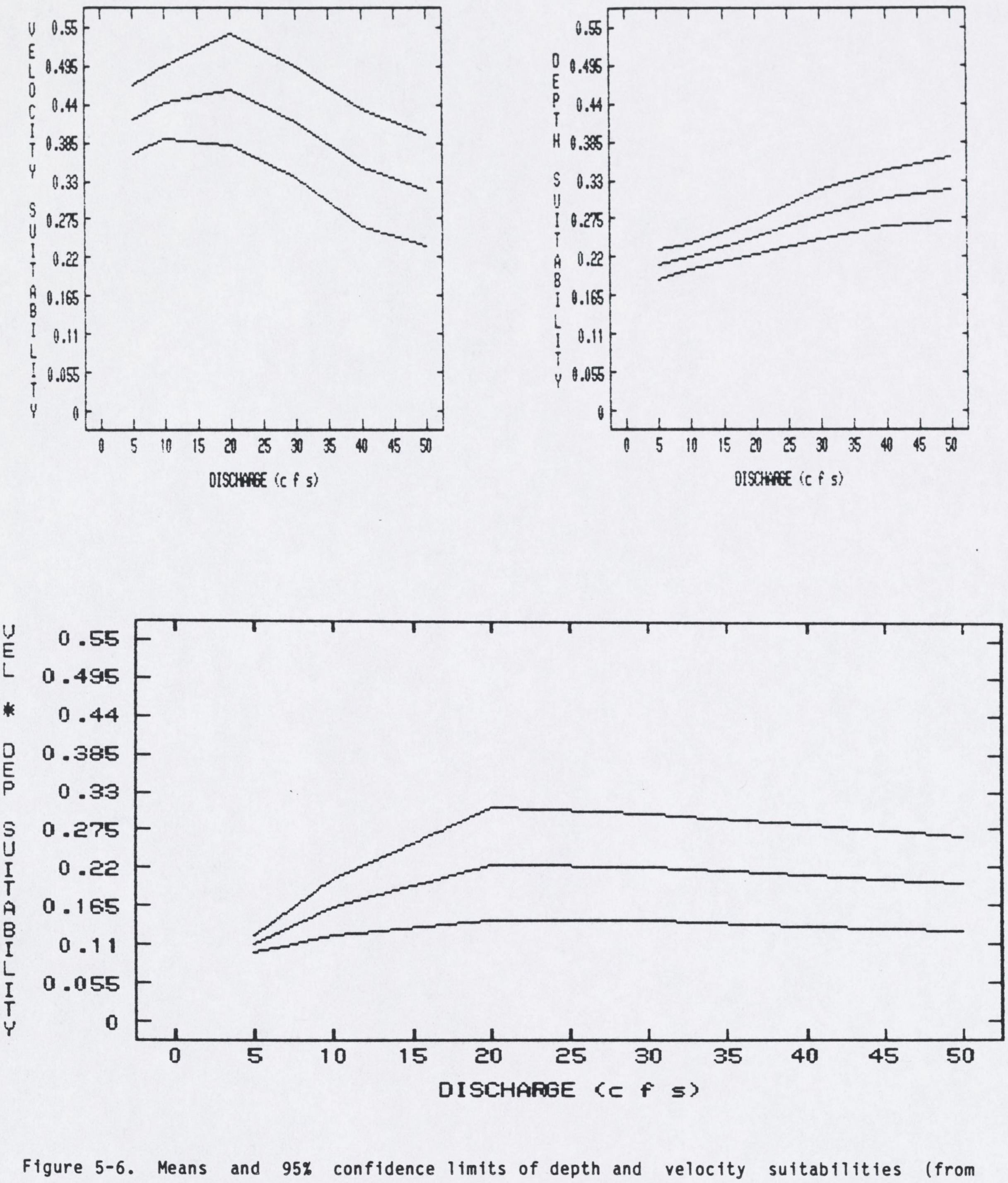
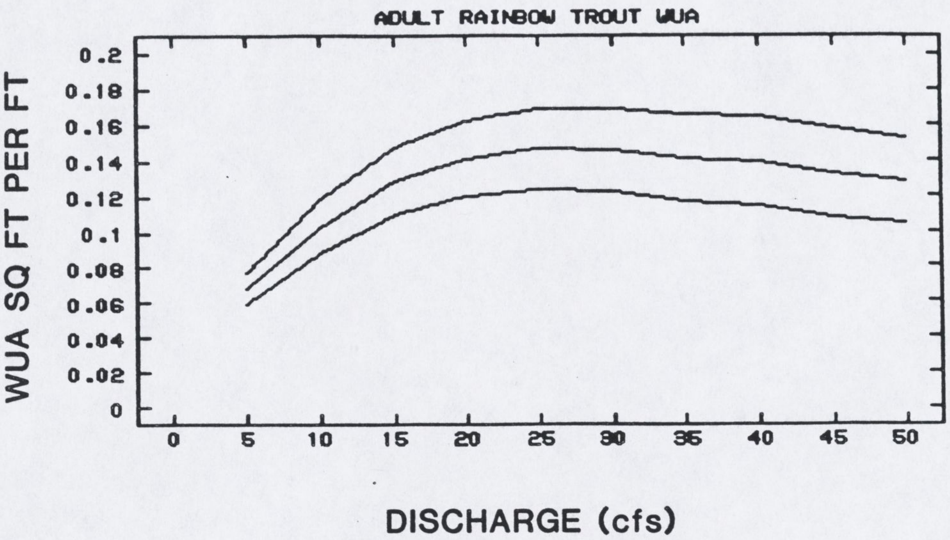


Figure 5-6. Means and 95% confidence limits of depth and velocity suitabilities (from Figures 5-4 and 5-5) and their product, showing that as a result of the shapes of the biological transformation curves, declines in velocity suitability are counter balanced by increases in depth suitability with increasing discharge. This effect is stream-specific.

Figure 5-7. Plot of Rainbow trout adult weighted usable area (ft^2 per linear ft of stream) showing mean and 95% confidence intervals. The confidence intervals shown are derived not from stochastic variation in the WUA, but from the distribution of depths and velocities at the measurement points along the transects. Rather than indicating the range within which one can be 95% confident of finding the true WUA, at a particular discharge, they show the range of WUA corresponding to 95% of the measurements.



at a large number of points along ten transects. Depth was varied ±10% and velocity was varied ±30%, which we thought was as large an amount of random error as was likely to occur in a field situation. The entire data set was subjected to this normally distributed random error 10 different times, and then run through the FWS IFG4 simulation model and the HABTAT model to produce 10 different plots of Weighted Usable Area versus discharge. At each discharge modeled, the 95% confidence interval was plotted and the results are shown in Figure 5-8. Even using large random errors, the 95% confidence intervals of Weighted Usable Area remained tight, and both upper and lower limits retained the shape of the mean curve. This stability is probably a direct result of the large number of input variables.

ERROR RESULTING FROM COMPUTER MODEL INCONSISTENCIES

not in refs.

Sanford (1984), in an intensive analysis of potential sources of error and their effects on the output of the IFG4/HABTAT model, notes that the IFG4 hydraulic model calculates discharge utilizing the standard USGS mid-section method in which "mean velocity and depth measurements are assumed to represent an area that extends halfway to the preceding and following verticals", but the HABTAT model uses the mean section approach in which "velocities from adjacent verticals are averaged to calculate a cell's average velocity and the product of mean velocity and cross-sectional area computed to obtain subsection discharge. Discharges computed by the mean-section method will always be equal to or less than those obtained by the mid-section technique (Savini and Bodhaine 1971)" and the discharges computed by the HABTAT and IFG4 models "often differ by as much as seven percent", resulting in potentially large errors in prediction of WUA.

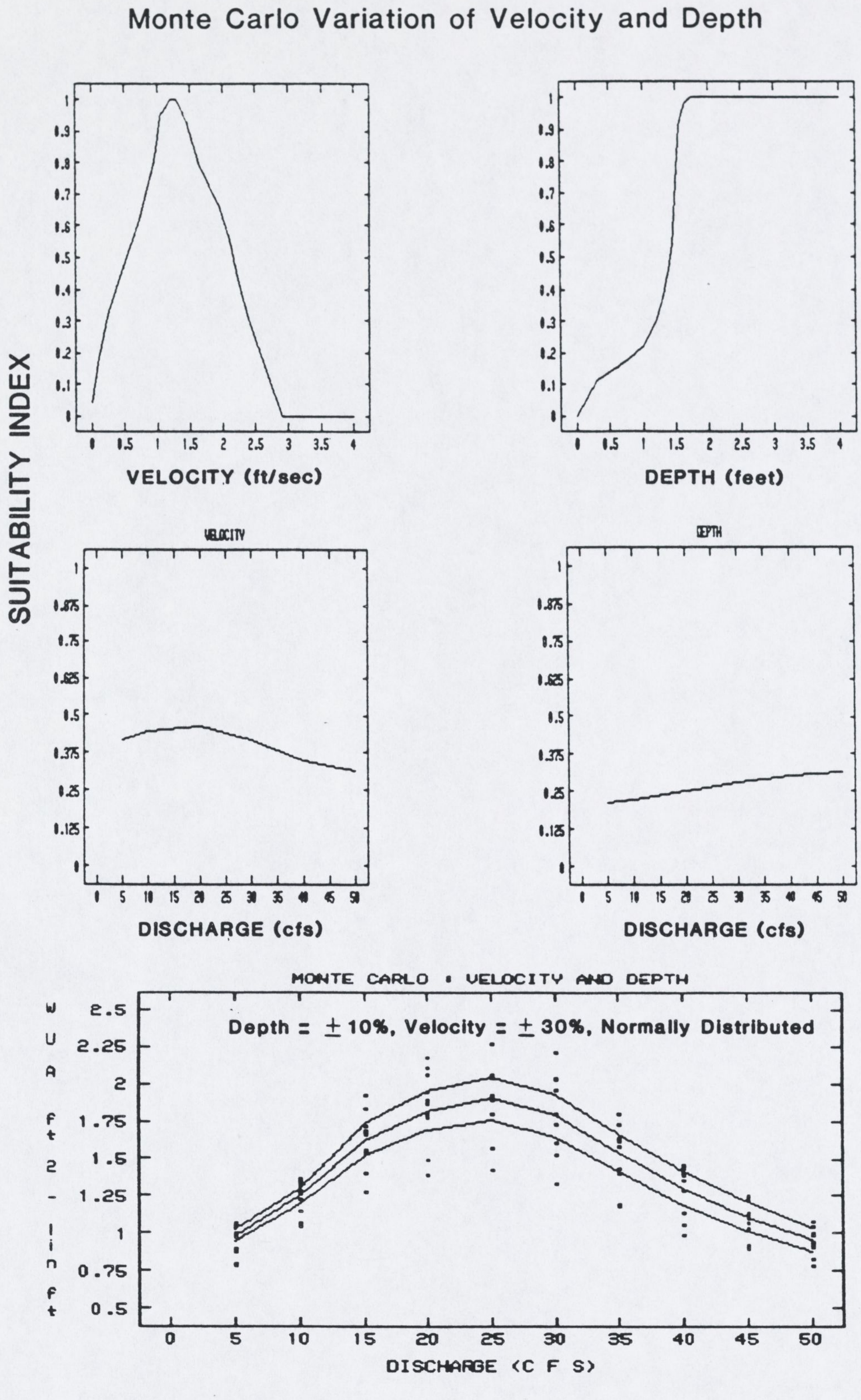


Figure 5-8. A Monte Carlo analysis of the effects of stochastic variation in velocity and depth on the Weighted Usable Area output of the IFG4/HABTAT model. The suitability indices used are shown at the top. The depth and velocity data that was varied is shown in the middle graphs (after IFG4 simulation), and the 95% confidence intervals of ten stochastic variations is shown at the bottom. The stability of the model is probably attributable to the large amount of input data.

CHAPTER 6: VALIDATION OF INSTREAM FLOW AND HABITAT QUALITY MODELS

There are several levels at which the validity of the models reviewed here can be examined. The most rigorous level is a test of whether a model accurately predicts a measurable response variable. The hydraulic simulation models and standing crop models are susceptable to such testing.

A second, less rigorous, level of validity testing consists of examining whether the response variable is correlated with, but not predictive of, a measurable response variable. Depending on the intended use of a model, this might be considered sufficient. For example, if the model output is a curve (such as wetted perimeter or Weighted Usable Area versus discharge) and the criterion for determining an appropriate instream flow is a relative point on the curve (the "inflection" point in the case of wetted perimeter, or the flow which maximizes habitat, or some other identifiable point, in the case of WUA), then a model might be judged valid if a measurable variable such as fish standing crop is found to be correlated with, but not predicted by, the response variable. But, it is very important to note that correlation does not imply causality. The presence of such correlations indicates that the model is descriptive of the standing crop, but does not show that changing a stream condition, such as discharge, would alter standing crop.

Randall & White varied Flow and measured SC. This was actually strongest test.

But regression really doesn't either, unless one expt. changes stream conditions and measures SC.

A third, much less rigorous, validation and one to which we hesitate to refer to as a validation at all, is the testing of the unmeasurable response variable of one model versus the unmeasurable response variable of another model. This is the approach that was used in the Methodology Evaluation Studies commissioned by the U.S. Fish and Wildlife Service to examine the relative oututs of the Montana Method (Tennant 1985), the IFG4/HABTAT and WSP/HABTAT methods, and any local or statewide methods also in use. Such a comparison does not get any closer to discovering the validity of any of the models, but may increase satisfaction with the simpler (less expensive) ones if their results are identical to the more complex ones. (These evaluation studies, it should be noted, were not intended as validation studies, and a major goal was specifically to compare costs of various models and to find out if the results of the simpler ones were similar to the more complex ones).

There has been a variety of model testing at all three of these levels, and this chapter is a review of the results.

VALIDITY OF MODELS USING BASIN VARIABLES

There are two reasons to incorporate or to rely entirely on basin-wide input variables in an instream flow model. The first is the lack of other information. All basin variables can be obtained directly from maps and are thus nearly always available in the absence of any other information about a stream. The assumption in using them as surrogates for stream-specific information is that the discharge and hydraulic characteristics of streams in a basin are entirely determined by basin-wide features such as drainage area, elevation, and gradient. It may turn out to be true in some places and if so, may make basin variables quite attractive. We have not seen any data supporting this concept, however.

The second reason to use them is that in models predicting measurable response variables such as standing crop, certain basin variables such as elevation, may be extremely important. Thus, using them as part of regression equations where they are conceptually appropriate as was done by Taylor (1980) in his riparian vegetation model and by Rabern (1984) in his fish population models, makes good sense and may be necessary to account for the variability among sites.

also Scarnecchia 1984.

There is not much evidence that any of these which? models are valid for predictive purposes.

VALIDITY OF MODELS USING DISCHARGE VARIABLES

Five of the models presented use a single discharge variable as the sole criterion for minimum instream flow, and they are all different. Annear and Conder (1974) advocate a maintenance flow consisting of either the mean annual flow or twice the mean annual flow less either the smallest statistically detectable difference from it or twice the smallest statistically detectable difference; Geer (1980) recommends the 6-month mean monthly flow for the period of record with separate recommendations for the summer and winter months; Larsen recommends the 25-year median daily flow; the NGPRP (1974) recommends the flow exceeded on 90 percent of the days in each month for the period of record excluding atypical months; and Tennant (1975) makes no recommendation, but implies that 30 percent of the average annual flow would be good. One other for which we did not produce a Methodology Summary Form, is Chiang and Johnson (1976), who recommended a flow equal to the lowest flow which lasts for seven consecutive days once every 10 years. None of the methods offer any real basis for their recommendations, so it is difficult to judge their relative merits or validity.

The greater the period of time over which the central tendency (mean or median) is determined, the less responsive the recommendation will be to seasonal fluctuations. Tennant's method using average annual flow and Larsen's 25-year median flow are therefore, much less likely to be correlated with natural seasonal flows than is Geer's 1980 use of six month mean minimum flows, which in turn is less likely to be correlated with natural seasonal flows than is the NGPRP (1974) use of monthly 90 percent exceedence flows. It is quite likely that some of these recommended flows will exceed the unregulated flows in some systems.

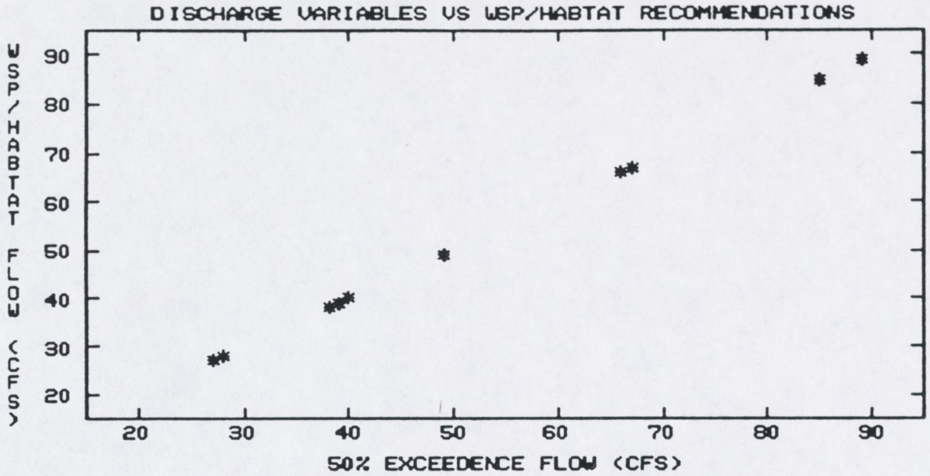
It is interesting, however, that the recommended flows resulting from using discharge variables alone may be similar to those using more complex methods. Hilgert (1981), for example, compared the results of using several discharge variables as criteria for monthly recommended flows and using the much more complicated USFWS WSP/HABTAT model. He obtained identical results using the median (50% exceedence) monthly flow and the flow he recommended (without explaining why) based on the WSP/-HABTAT method (Figure 6-1). The result was characteristic of most of the streams he studied. We also plotted the 90% exceedence flow (the NGPRP 1974 criterion) and mean monthly flow (30% of which would be conceptually similar to Tennant's criterion) against the WSP/HABTAT output with much less good correlations (Figure 6-1).

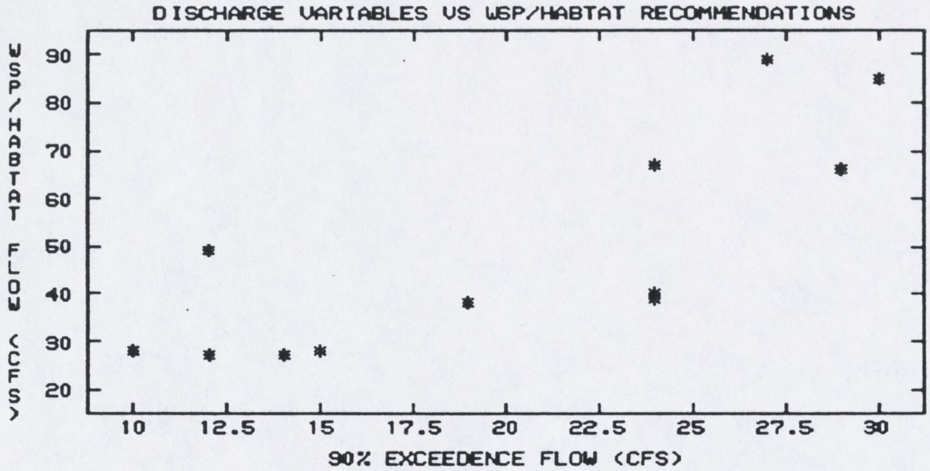
Horton and Cochnauer (1980), on the other hand, interpreted their WSP/HABTAT data to recommend flows that are identical to 30 percent of the mean annual flow (the usual criterion of the Tennant Method) (Figure 6-2). Interestingly, use of the WSP hydraulic model instead of the IFG4 hydraulic model, produced recommended flows strongly correlated with, but about half the absolute value of the 30% mean annual flow. This is a particularly surprising result, since the IFG4 and WSP hydraulic simulation models are simulating the same thing and should produce identical results.

One might conclude from these two data sets that there is nothing to be gained from using the IFG4/HABTAT or WSP/HABTAT, instead using a simple discharge criterion. But since neither Hilgert nor Horton and Cochnauer describe how they converted the output of the IFG4/WSP/HABTAT models to recommended flows, and since the discharge statistics used by Hilgert (median monthly flow) was quite different from that used by Horton and Cochnauer (30% mean annual flow) and the results with the IFG4/HABTAT model were different from those using the WSP/HABTAT model in both studies, it is difficult to know how to decide which of the criteria is the appropriate one and how to implement it.

In a similar vein, Nehring (1979) compared Tennant's 30 percent average annual flow criterion with a set of hydraulic data weighted with binary mean depth and velocity criteria. The WSP and IFG4 models were used to generate the mean depth and velocity data. The criteria used and the results are shown in Figure 6-3. The recommended flows using either model are virtually identical to the results using the 30% average daily flow criteria alone.

This result is interesting because it shows that in the 13 streams sampled by Nehring in Colorado, mean depth and velocity are strongly correlated with mean annual discharge, and what's more, mean depths and velocities meeting Nehring's biological criteria occur at exactly 30% of mean annual discharge. This reflects Tennant's general concept that historical discharges





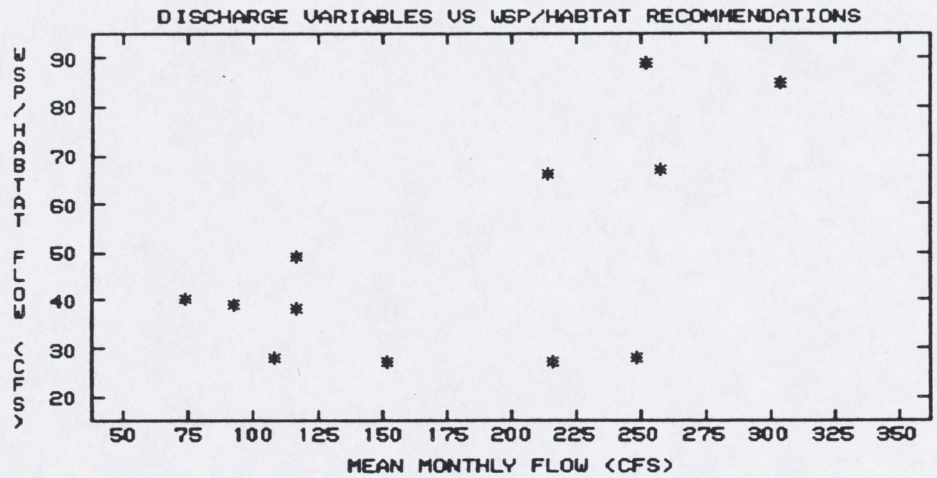
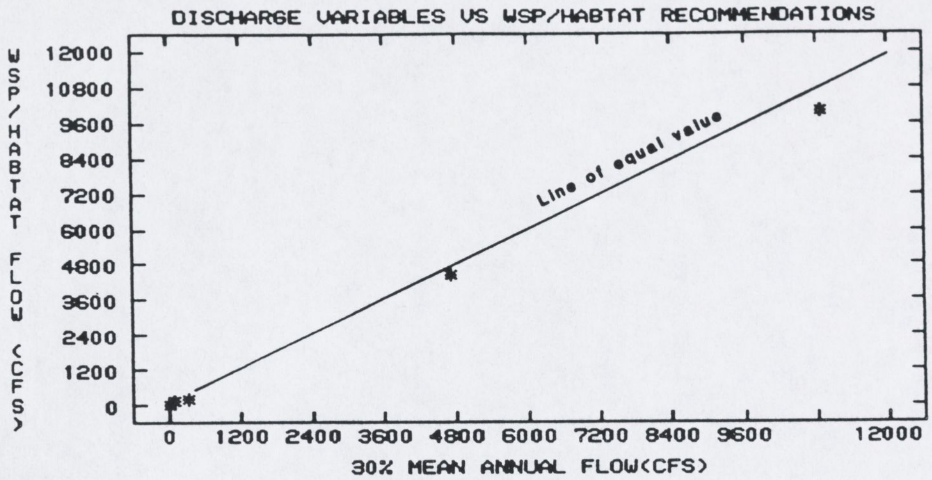


Figure 6-1. Correlations between discharge variables (50% monthly exceedence flows = median monthly flows, 90% monthly exceedence flows, and mean monthly flow), and the recommended flow derived from the FWS WSP/HABTAT method. Data from Hilgert (1981) for the North Fork Big Nemaha River, Nebraska. Hilgert got similar results for other streams as well, but does not explain how he arrived at the WSP/HABTAT recommended flows.



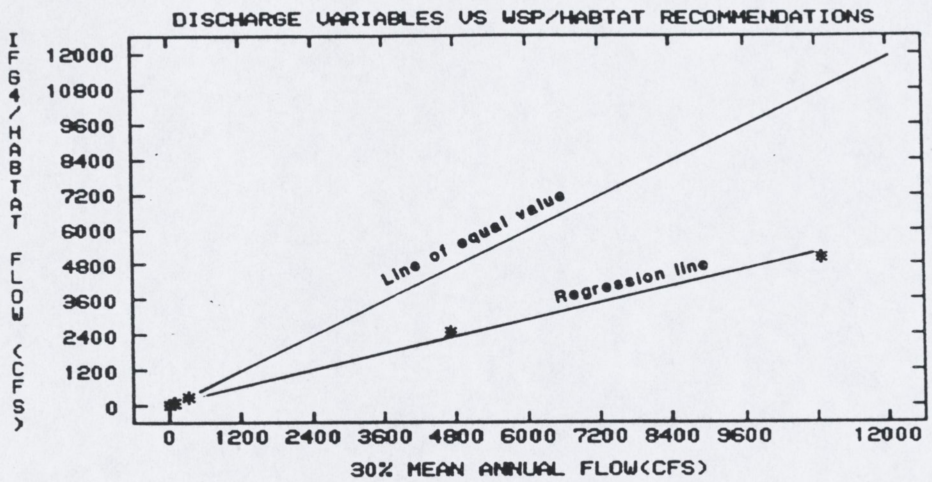
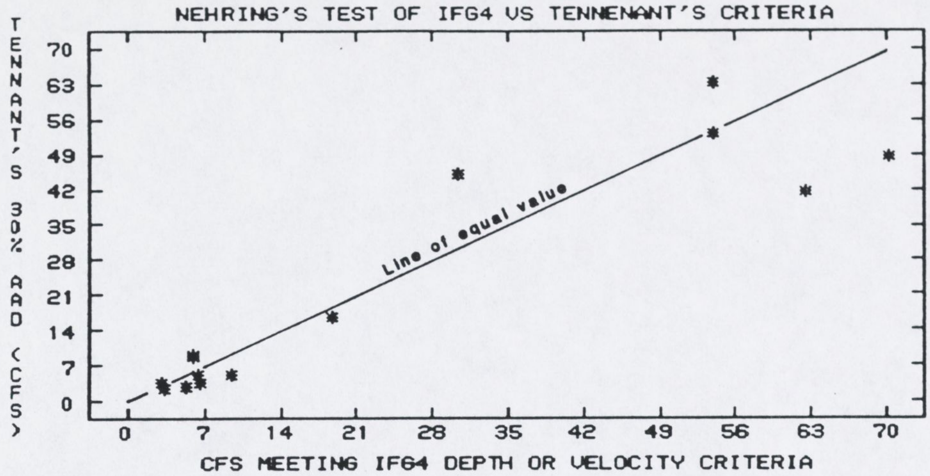
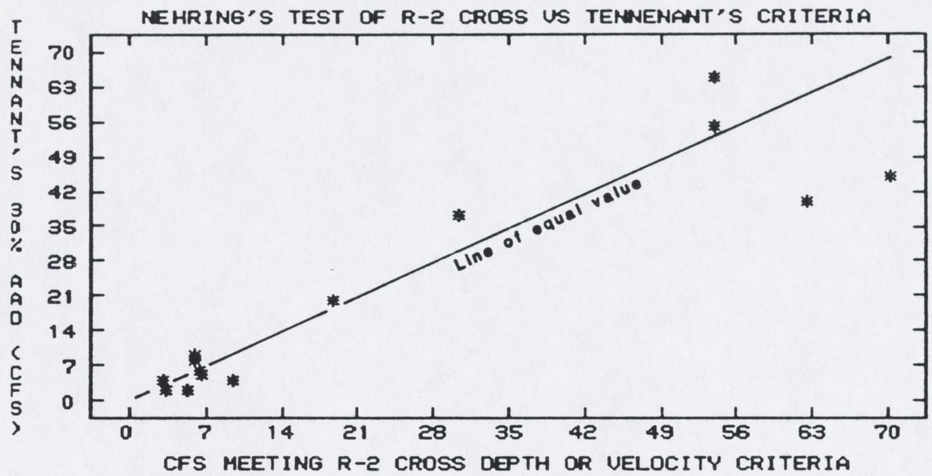


Figure 6-2. Comparison of flows recommended by the Tennant method (30% mean annual flow) and mean annual flows recommended by WSP/HABTAT and IFG4/HABTAT methods (data from Horton and Cochnauer 1980). Note the strong correlations and the differences between the WSP and IFG4 (theoretically identical) models.





Stream width (ft)	Average depth (ft)	Average velocity (ft/sec)
	Run Habitat Type	
1 - 21	0.2	0.5
21 - 40	0.2 - 0.4	0.5
41 - 60	0.4 - 0.6	0.5
61 - 100	0.6 - 1.0	0.5
	Pool Habitat Type	
1 - 20	0.4	0.1
21 - 40	0.4 - 0.8	0.1
41 - 60	0.8 - 1.2	0.1
61 - 100	1.2 - 2.0	0.1

Figure 6-3. A comparison of instream flow recommendations based on Tennant's (1975) 30% Average Annual Flow Criteria and the binary depth and velocity criteria (listed above) derived from R-2 Cross and IFG4 Hydraulic models. The HABTAT model was not used. (Data from Nehring, 1974).

shape a stream channel in a way that causes a fixed percent of mean annual discharge to provide optimum depths and velocities.

VALIDITY OF MODELS USING UNTRANSFORMED HYDRAULIC VARIABLES

We have identified two types of untransformed hydraulic variables: those that influence organisms only indirectly by being correlated with the size of the stream (e.g., width, toe-of-bank width, wetted perimeter) and those which influence where an organism might choose to be (e.g., depth, velocity).

Absolute stream width is likely to have some importance in determining the standing crop of organisms, simply because habitat availability and food production are likely to increase with width, other things being equal.

Nehring (1979), for example, has data with a strong correlation between stream width and standing crop (Figure 6-4). Wesche's (1980) Brown trout data plotted against the mean width (determined by taking the average of the range supplied by him) do not show as good a relationship, however (Figure 6-4). Recent data (McEwen and Deinstadt, unpublished) show a correlation, but explain only 20% of the variation in standing crop.

not in refs

Wetted perimeter is an attractive index of habitat, because its putative relationship to invertebrate production. Cada et al. (1983), however, examined the hypothesis that wetted perimeter should reflect the benthic food resource available to support stream fishes under varying flows, and concluded that "the modeling of discharge/wetted perimeter dynamics does not provide adequate understanding of biotic interactions needed to develop sound instream flow recommendations." The reason for the conclusion was that as discharge and wetted perimeter decreased, the absolute invertebrate biomass <u>increased</u> as shown in Figure 6-5 The discharges did not get low enough for wetted perimeter to decline drastically and it might be argued that the "inflection point had not been reached and therefore, that no decrease in invertebrate density would have been expected.

Randolph & R.G. White 1984.

Randolph (1984) conducted some ingenious studies in which he monitored the emigration of rainbow trout from their test sections of stream as flows gradually decreased. The intention was to test the validity of the wetted perimeter method for recommending instream flows, and indeed the number of fish remaining decreased as discharge decreased (and coincidentally, as wetted perimeter decreased) (Figure 6-6). The association with an identifiable point on the wetted perimeter versus flow curve was less clear, however, and if anything, suggested that the abrupt change in the wetted perimeter curve occurs at a lower flow than is optimal for trout.

? how could wet P ↑ as Q ↓ ?

The other type of hydraulic variables are generally dominated by depth and velocity, which have become extremely important in the

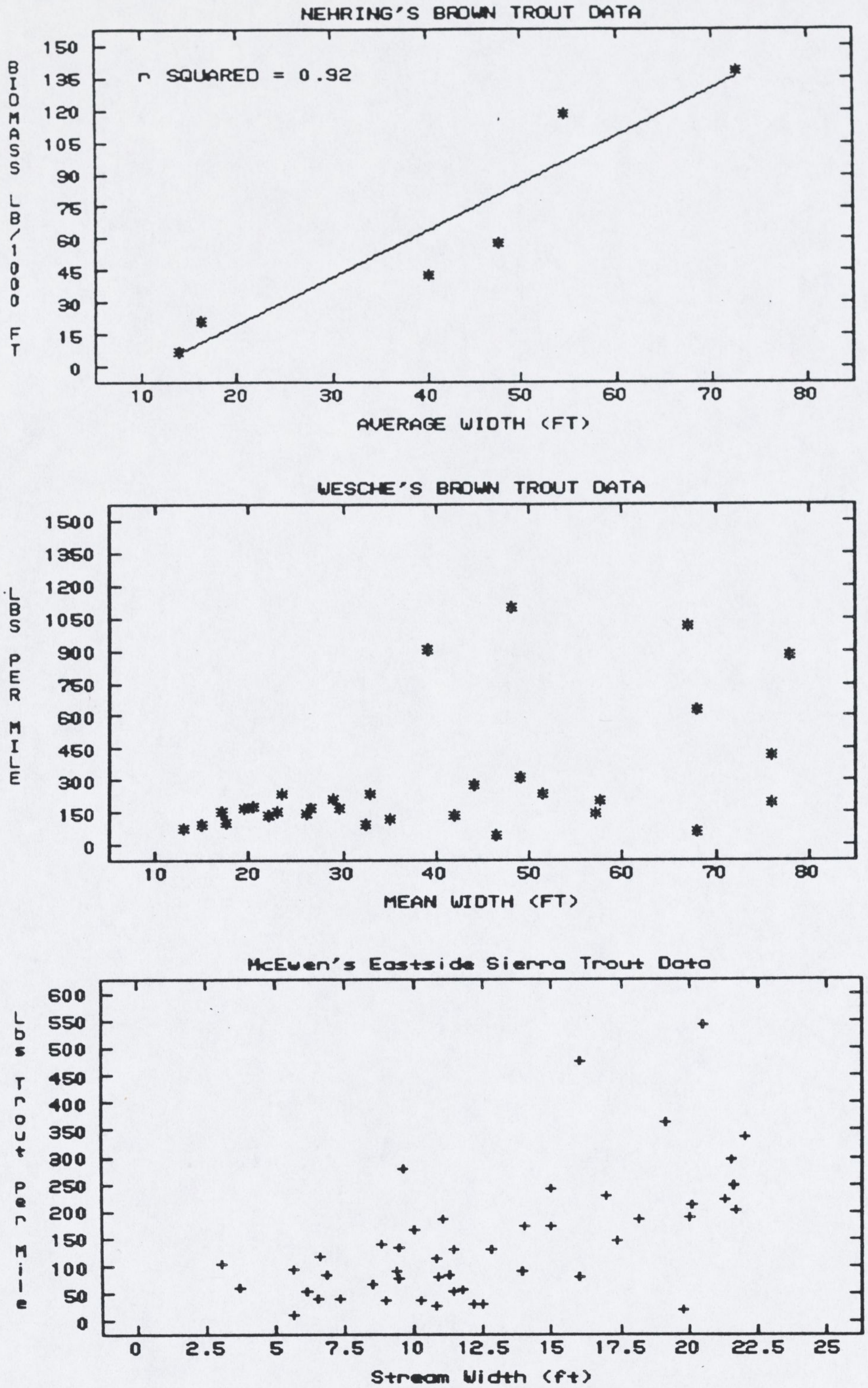


Figure 6-4. Standing crop as a function of stream width. Data from Nehring (1979), Wesche (1980), and McEwen and Deinstadt (unpublished).

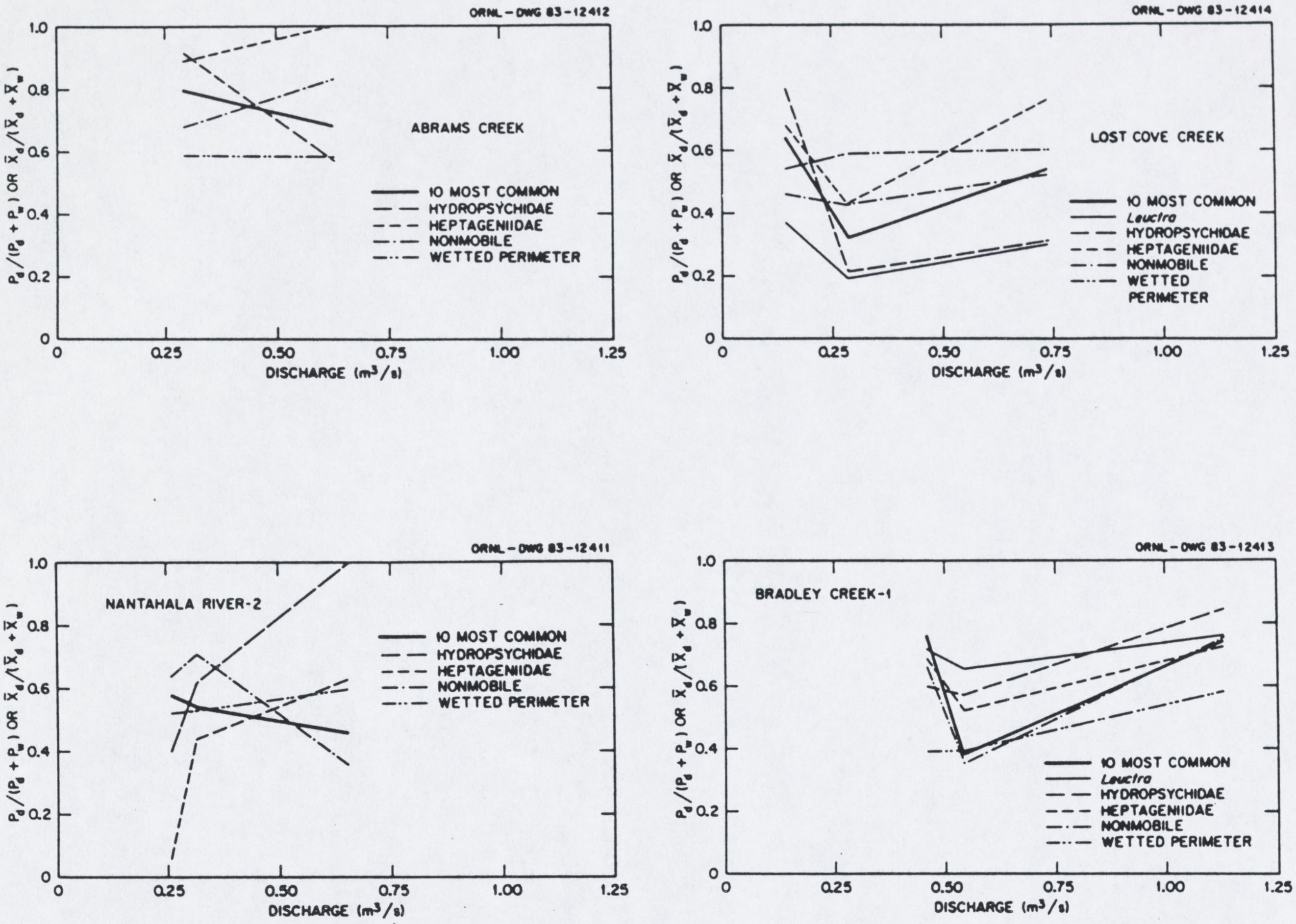


Figure 6-5. Cada et al.'s (1983) data showing that as discharge and wetted perimeter decrease, benthic invertebrate biomass sometimes increases. Both wetted perimeter and invertebrate biomass are in dimensionless units roughly comparable to the decimal fraction of their values of bankfall discharge. (Figures copied from Cada et al. 1983).

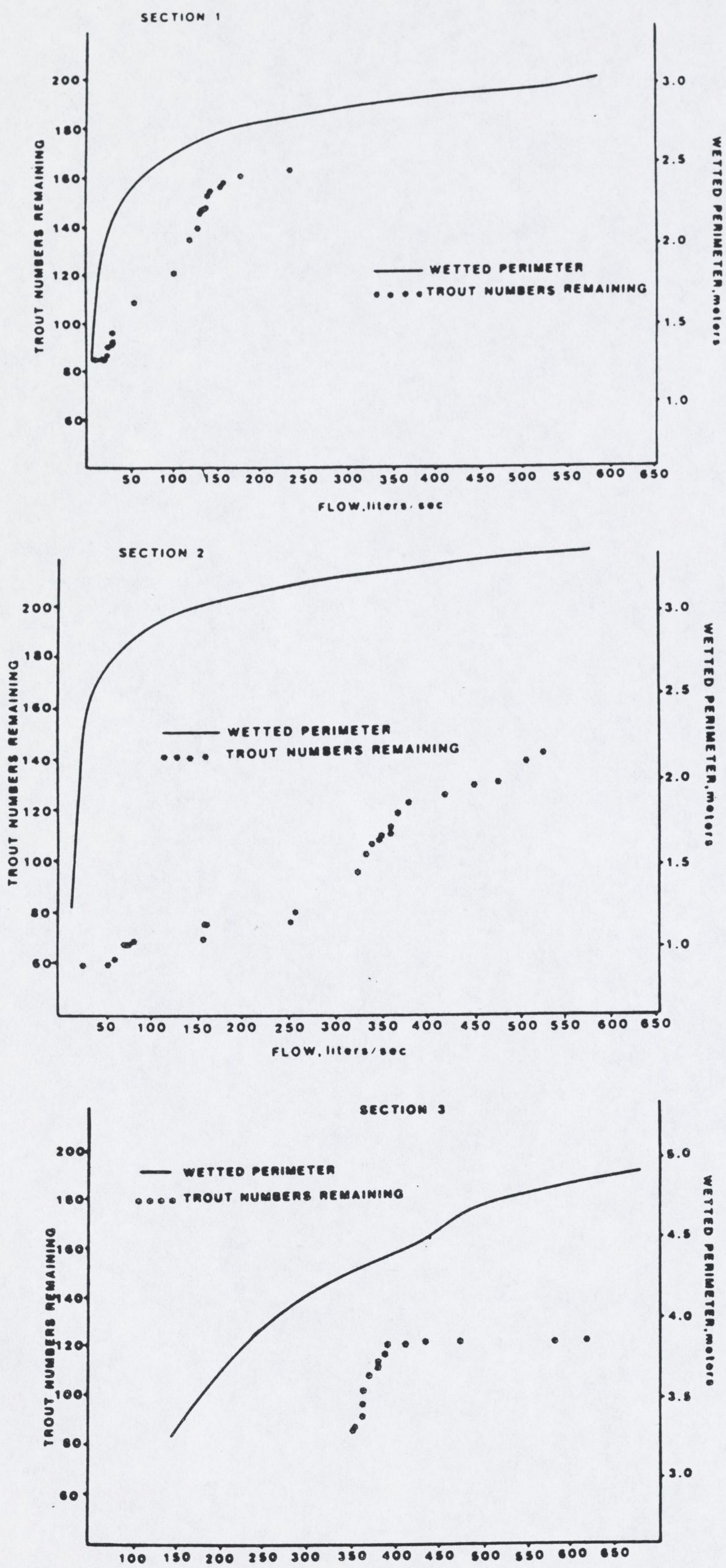


Figure 6-6. The relationship between wetted perimeter, number of trout remaining, and flow in 3 sections of Ruby Creek, Montana (from Randolph, 1984). Note that although in all sections there is a rather abrupt decrease in the number of trout as flow diminishes, it would be difficult to identify a point on the wetted perimeter curve corresponding to this point.

development of instream flow modelling. The FWS HABTAT model effectively uses only depth and velocity in most applications (since the usual third variable, substrate, tends not to change with flow), and these terms appear in many of the other models reviewed as well. ?

It is quite clear that many fish tend to occupy water within certain ranges of depth and velocity, and that this can be seen either by gridding off an area and observing the number of fish present as a function of depth and velocity (see Figure 6-7 for an example of a typical data set), or by observing the location of fish and measuring the depths and velocities occurring where the fish are (see Figure 6-8 for a similar example of this kind of data).

It is also clear that the distributions of mean column velocity and depth change with discharge. The general pattern of change occurring in a small stream is shown in Figures 6-9 and 6-10. At low discharges, both depths and velocities tend to be changed at low levels. As discharges increase, an increasingly even distribution of depths and mean column velocities occurs.

What is less clear is the extent to which fish populations are limited by the presence of appropriate depths and velocities, and in the case of velocity, the appropriate place to measure it. Jenkins (1969), in an intensive study of the behavioral selection of locations for feeding by brown trout, suggested that the choice of velocities was based not on mean column velocity, but on the proximity to surface flow patterns that maximized food delivery through drift. It appears from his data that relative rather than absolute velocities (within levels that can be withstood by fish) are important, and that the fish could withstand higher mean column velocities than they preferred by finding velocity refuges either in bottom depressions or associated with rocks. This same phenomenon has been noted by Chapman and Bjornn (1969), Everett and Chapman (1972), and Griffith (1972). Fausch and White (1981) assumed that positions with the greatest difference in water velocity between the focal point of brook and brown trout and within 60 cm of the focal point, were proven advantageous. All of these observations suggest that the mean column velocity may be completely irrelevent to many fish (except in situations where it is correlated with some other velocity of interest) and that the nose velocity (the velocity at the fishes nose) is only important if it becomes so high that the fish wastes energy trying to maintain position.

In view of the apparent preference for relative rather than absolute velocities, at least for territorial fish dependent on drift, it seems likely that the apparent preferred mean column velocity would be a function of discharge, since the relative velocites at a given location could remain similar even though the absolute velocities varied. We have seen no work exploring this possibility, however.

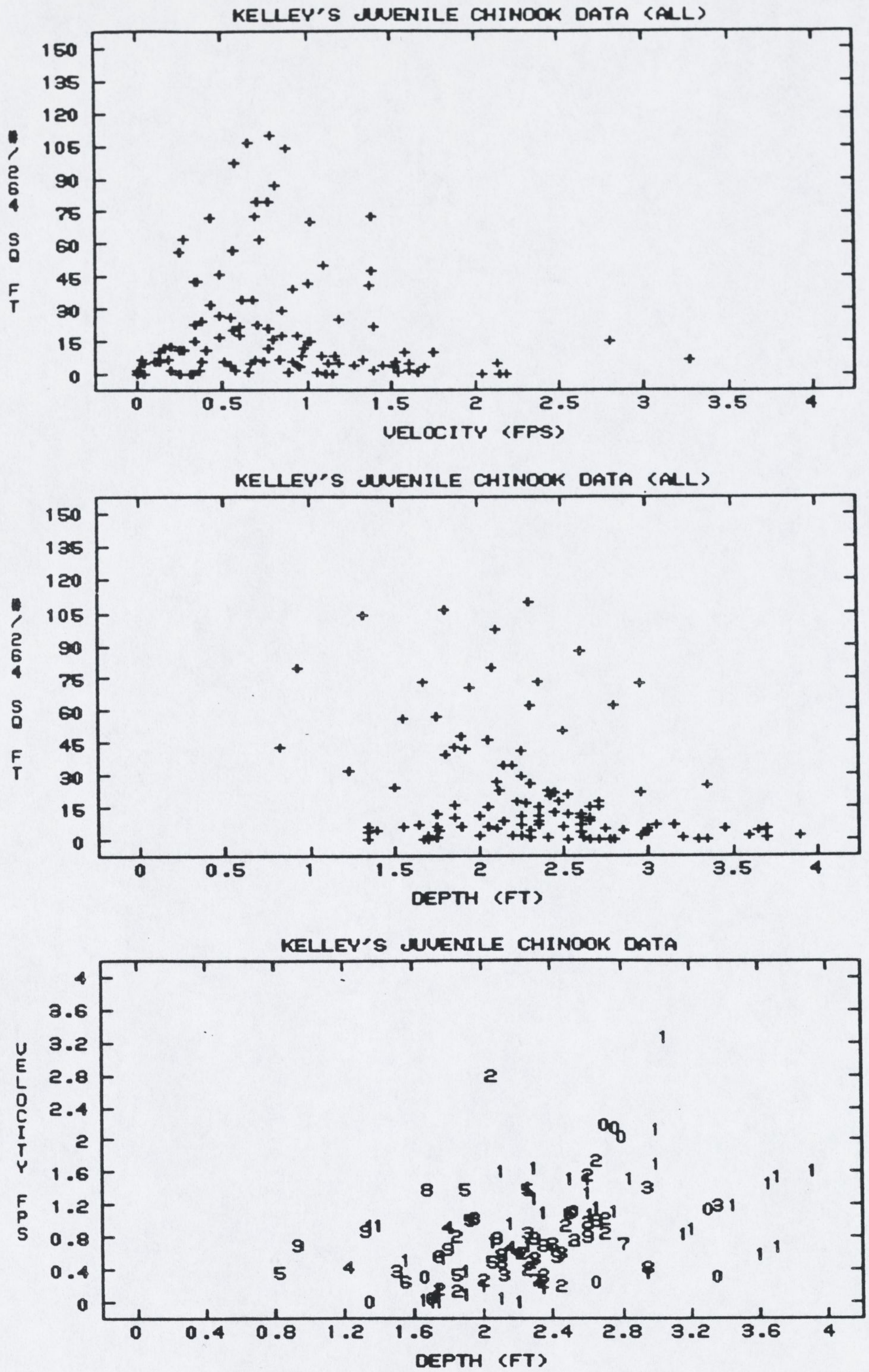


Figure 6-7. The relationship between number of juvenile salmon present in the lower American River (California) as functions of velocity and depth (data from Kelley et al. 1985). Each data point represents the number of salmon present in a 3 ft by 5 ft grid cell of several grids established in the river. The bottom figure shows numbers of juveniles per grid cell (1 = <10, 2 = 10-19, 3 = 20-29, etc.) as a joint function of depth and velocity. Note that very few fish occurred at depths >2.8 ft or velocities >1.6 fps, even though those conditions existed in many grid cells. These data do not provide information on depths <0.8 ft, since no grid cells were located in water that shallow.

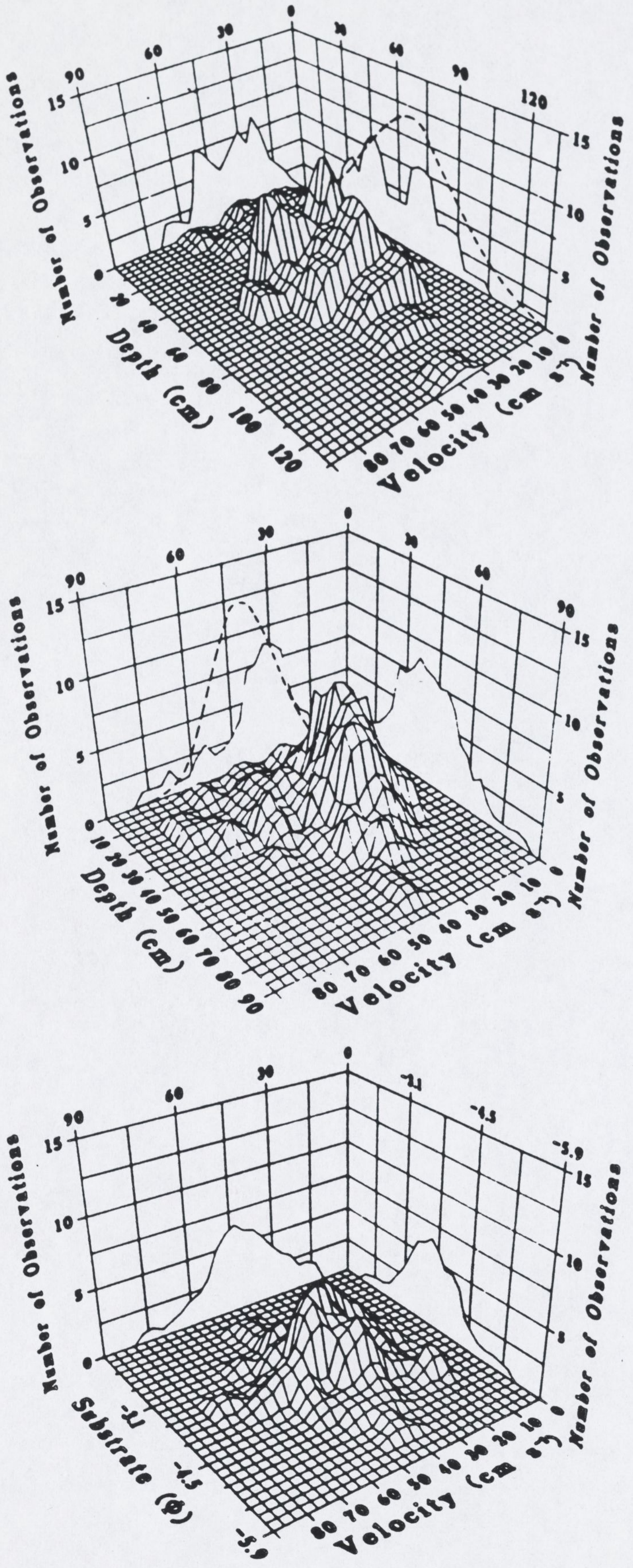


Figure 6-8. A data set based on observing depths and velocities at the point where brown trout were feeding (top illustration) or spawning (bottom illustrations) in New Zealand rivers (from Shirvell and Dungey 1983). Note that the depths, velocities, and substrate types selected by the trout are limited, and were shown by Shervell and Dungey to be a subset of those available.

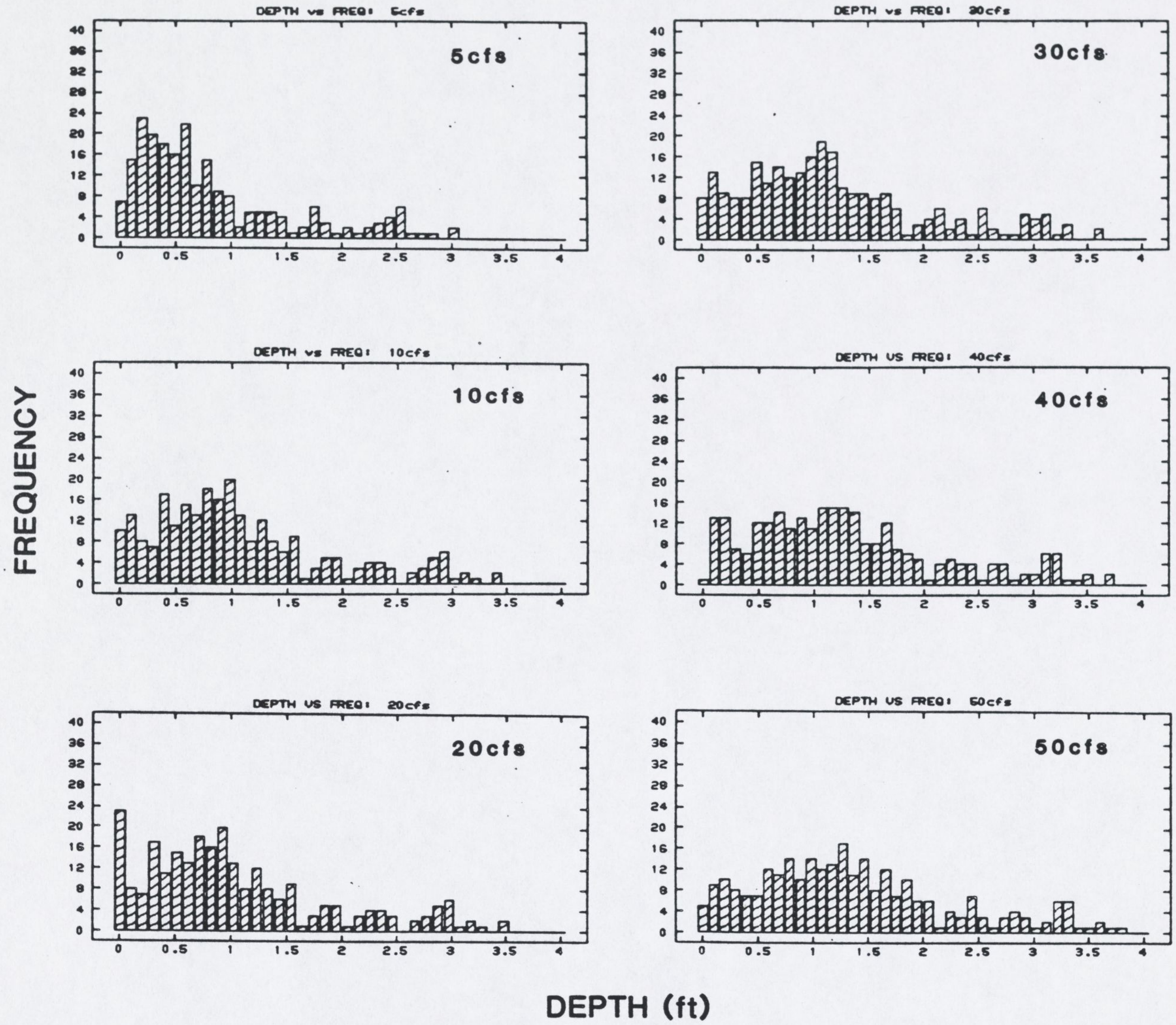


Figure 6-9. Frequency distribution of depth at transect measurement points at discharges from 5 to 50 cfs. The data are from a small Sierran stream and show a tendency for mean depth to increase and the distribution to become more even as discharge increases. (Original data collected by EA. These results are based on a simulation using the IFG4 hydraulic model).

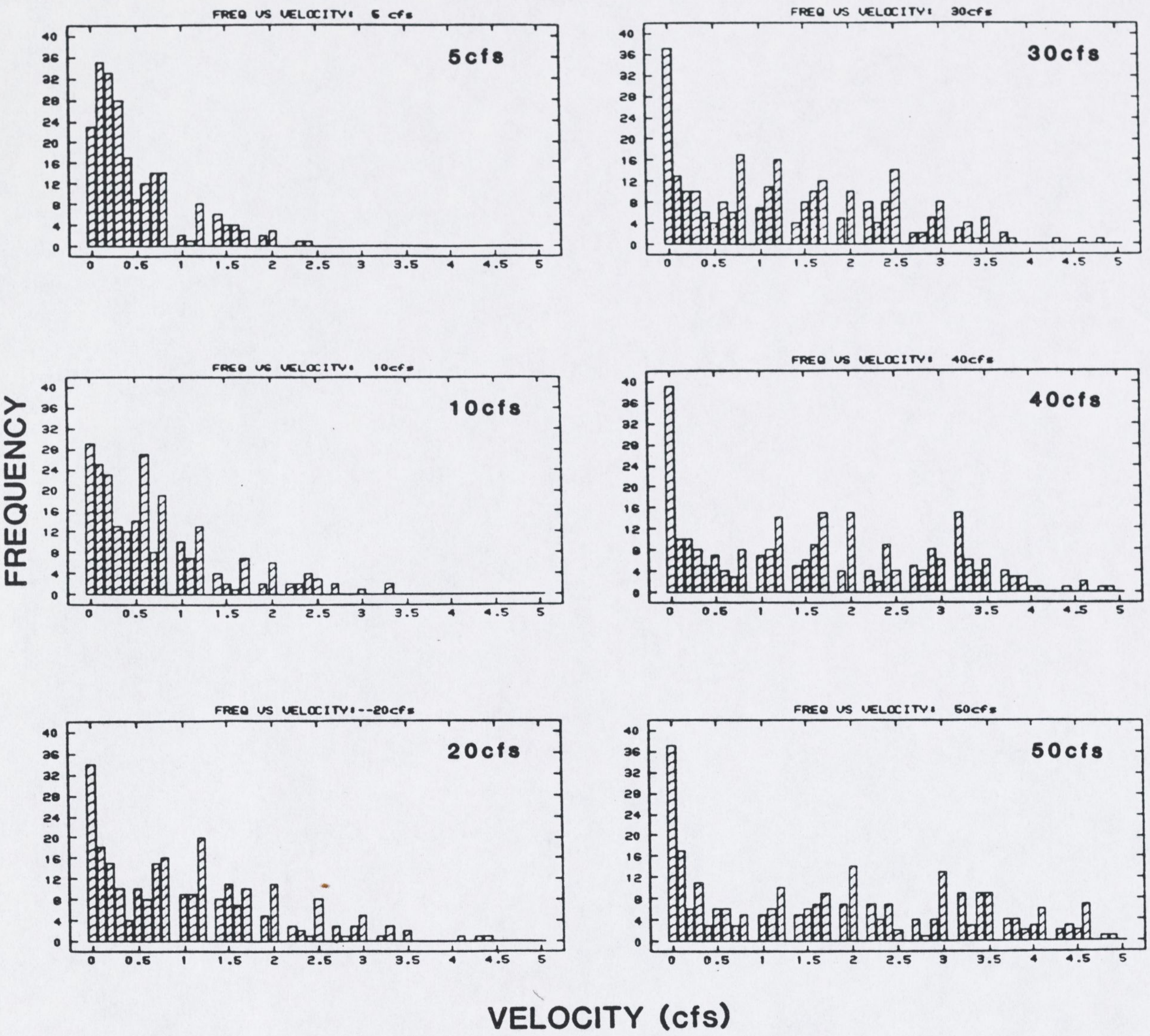


Figure 6-10. Frequency distribution of velocity at transect measurement points at discharges from 5 to 50 cfs. The data are from a small Sierran stream and show the tendency for the velocities to become more evenly distributed as discharge increases. (Original data collected by EA. These results are based on a simulation using the IFG4 hydraulic model).

We also expected that there might be good data available showing the relationship between standing crop or some measure of fish production either as functions of depth and velocity in undiverted streams, or, ideally as functions of the changed distribution of depths and velocities in diverted streams. Most of the standing crop models for undiverted streams used depth or velocity or some combination of the two as variables (Barber et al. 1980, Binns and Eisermann 1979, Li et al. unpublished, Nickelson et al. 1979), but none of them examined the relationships in streams which had been diverted.

seems data from Randolph and White should fit.

?

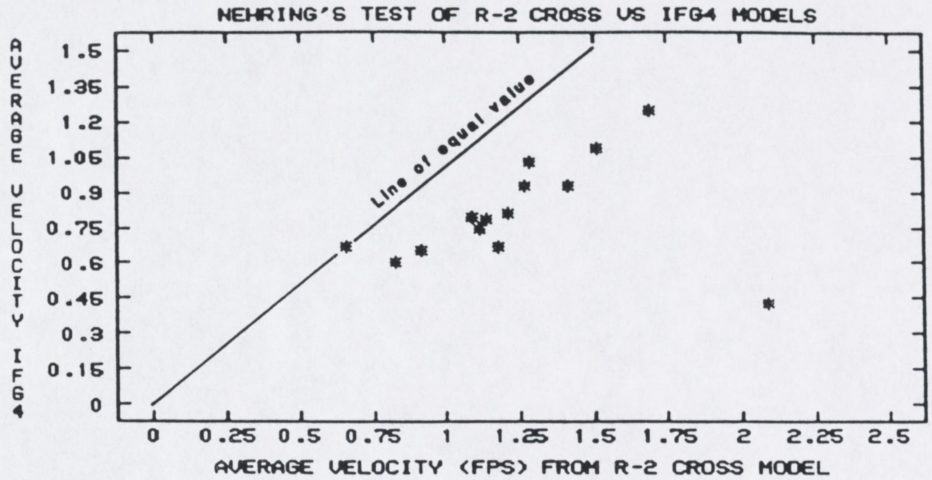
VALIDITY OF HYDRAULIC MODELS SIMULATING HYDRAULIC VARIABLES

There have been two kinds of testing (some of it inadvertent) of the three hydraulic simulation models reviewed here. Most of the tests have been comparisons of the results produced by two hydraulic simulation models, but there has been some specific testing for validity.

The only study we found making intentional comparisons between hydraulic models was Nehring (1979), in which the R2-Cross and IFG4 models were tested for their ability to predict average depth and average velocity. Figure 6-11 illustrates the results of one of Nehring's tests demonstrating that the IFG4 model produces estimates of average channel depth from cross sectional data higher than those produced by the R2-Cross model, but estimates of velocity which are lower. It is surprising that they do not produce identical estimates of average depths, since (presumably) the same depth data along transects were entered into each model and all either model has to do is add them up. Nehring does not explain the reason for the discrepancy, which is in some instances quite large. Some of the depth estimates produced by the IFG4 model were twice as large as those produced by the R-2 Cross model. It is somewhat comforting that the results of both models are at least correlated. The R-2 Cross model predicts velocities which are systematically much higher than those produced by the IFG4 model. Unfortunately, although Nehring examined the question, it is not clear which of the models produces the correct result.

The same type of results demonstrating a lack of comparability between the results of the WSP and IFG4 hydraulic models was obtained by Butler (1979). Butler did not compare the hydraulic output of the two models, but did produce tables of the percent maximum Weighted Usable Area as a function of discharge for 12 species of fish, using both the IFG4/HABTAT and WSP/HABTAT models. We plotted the data (Figure 6-12) and found some striking differences in the shapes of the curves. Hilgert (1979) got similar results from a similar comparison.

Although neither author mentions these discrepancies, the implications of their results for instream flow models dependent on hydraulic simulation are profound. What these comparisons



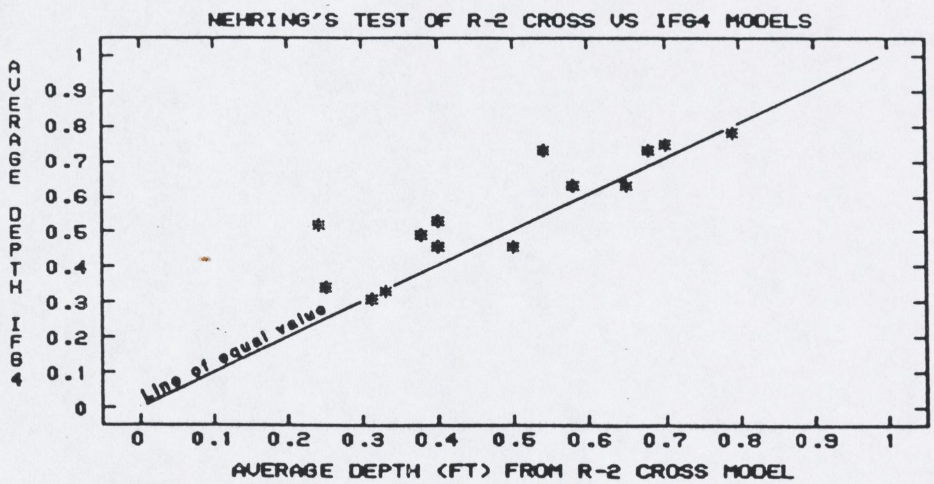


Figure 6-11. Comparison of average transect depths calculated using the R-2 Cross and IFG4 Hydraulic models. The IFG4 model calculates velocities lower than does the R-2 Cross model, but slightly higher depths (data from Nehring 1979).

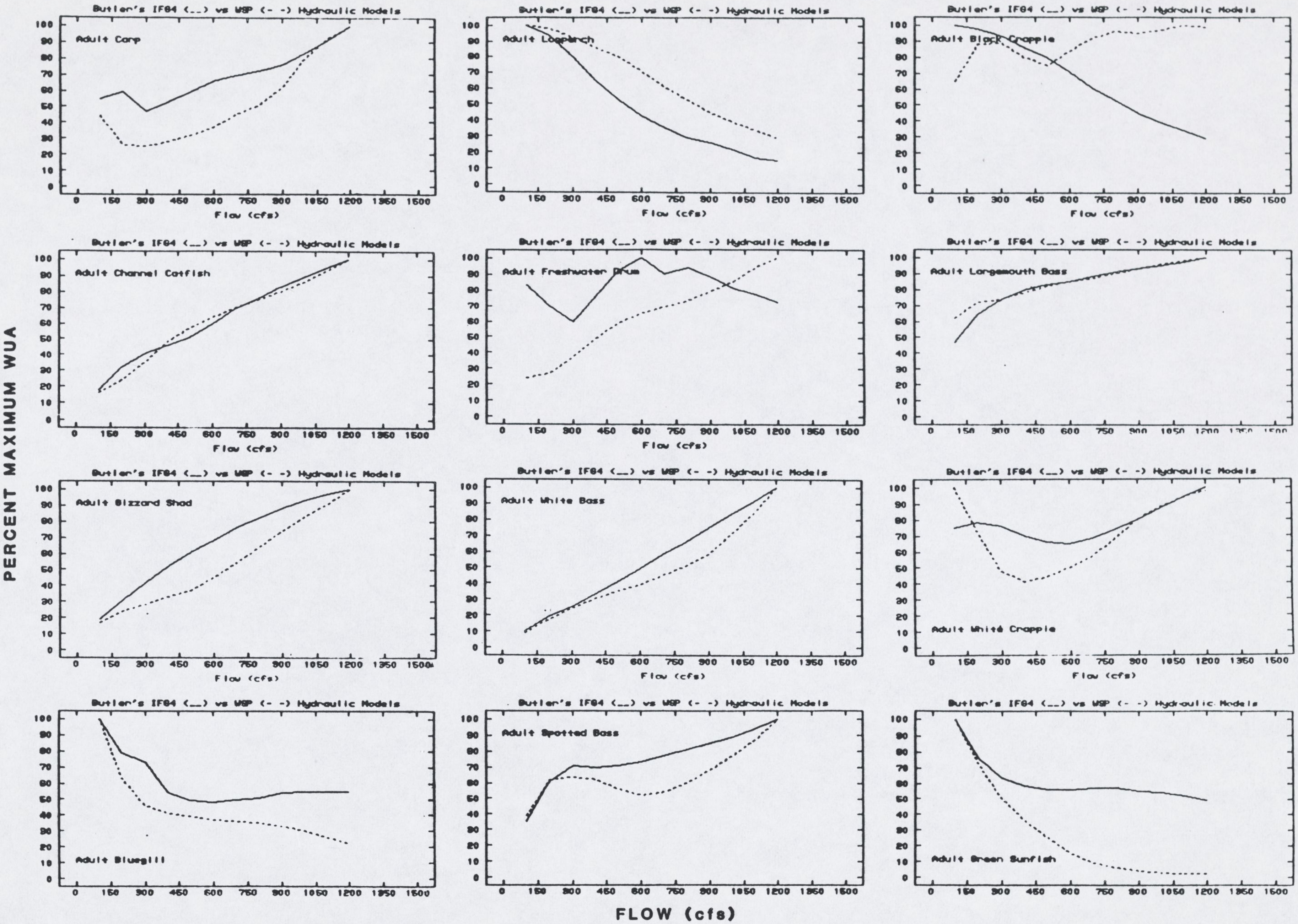


Figure 6-12. Comparison of WUA vs flow calculated using IFG4 and WSP hydraulic models for the Brazos River (Butler 1979)

show is that at least some of the hydraulic simulations, which are not usually questioned, are not accurate and should not be relied on for instream flow modeling. The problem is exacerbated by the fact that all three of the hydraulic models reviewed can be expected to perform best in situations where turbulence and complexity of bed structure are minimized, and where gradients are not high.

VALIDITY OF MODELS USING WEIGHTED HYDRAULIC VARIABLES

All of the validation tests we have discovered in this class are directed at the IFG4/HABTAT model. We have include figures in this section which represent all the experimental results we have located.

Figure 6-13a (from Stalnaker 1979) shows one of the original data sets used to demonstrate the relationship between Weighted Usable Area and biomass. Notice that the pattern formed by the data points is not closely approximated by the regression line, and the data set could be equally well interpreted as showing no relationship at WUA units less than 20. The units used for expressing WUA (per unit area or per length of stream) were not included in the report. ✓

Nelson (1980) produced plots of pounds of rainbow and brown trout per mile versus WUA (Figure 6-13b). There appears to be a much stronger relationship between rainbow trout biomass and WUA than between brown trout biomass and WUA. No units (per area or per mile) were given for the WUA values. get

Wesche (1980) presented brown trout data (which appear to be a subset of the data used by Stalnaker (see Figure 6-13a) showing the WUA expressed as square foot per linear foot of stream versus biomass. He also distinguished between small (average daily flow <100 cfs) and large streams. We plotted these WUA data, segregated by stream size against biomass expressed both as kg/ha and as kg/km (Figure 6-14). In both cases, there was no relationship between WUA and biomass in the small streams (as suggested also by Stalnaker's plots), but there was a strong relationship when biomass was expressed per km and only the data from large streams were considered. We also plotted Wesche's results for brook trout (Figures 6-15a,b), for which there was no correlation at all between WUA and standing crop. The WUA values for Wesche's data were determined for the discharges at which the biomass was sampled and there appeared to be no attempt to correlate biomass with extremes of WUA occurring in the stream.

Nehring (1979) calculated WUA that occurs at Average Annual Flows and measured biomass for both brook trout and brown trout. The brook trout data are shown in Figure 6-16a. Nehring saw no relationship between the brook trout standing crop and WUA, but the data were fit very nicely with a 4th order polynomial, so we added it to the plot. Nehring also reported stream depth and So?

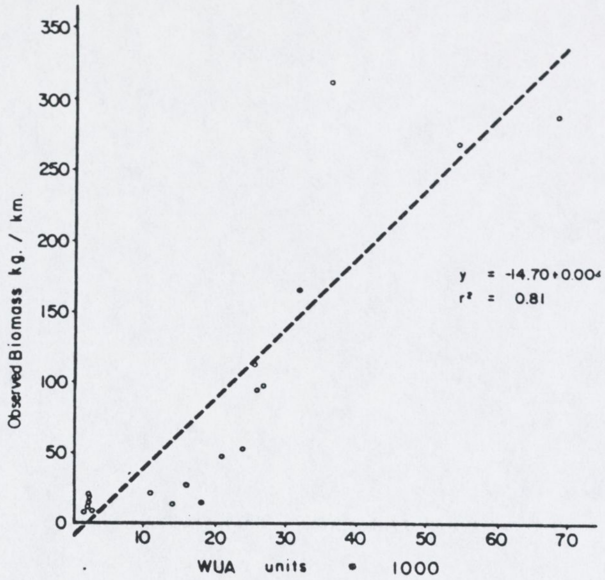


Figure 6-13a. The first data set used to demonstrate a correlation between Weighted Usable Area and biomass, based on data provided by Wesche (1976) and reported in Stalnaker (1979).

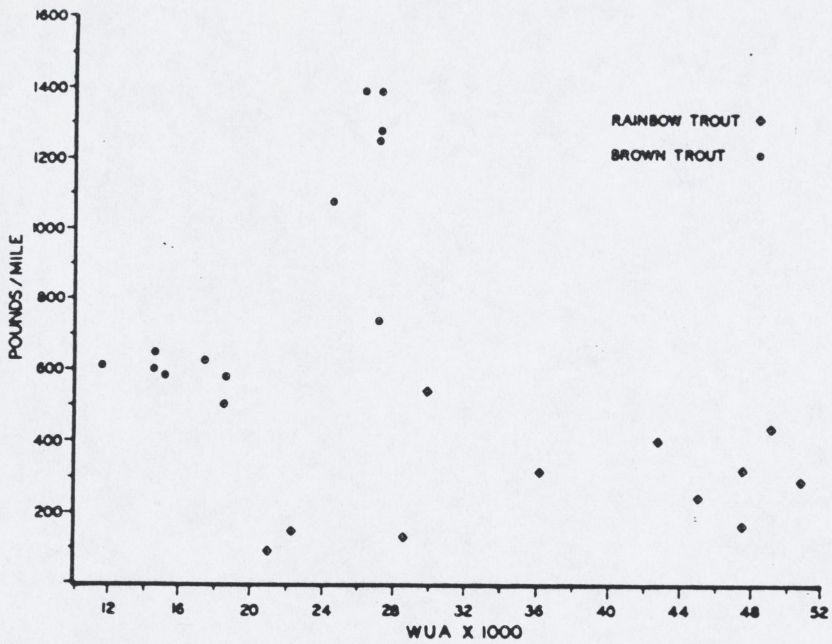
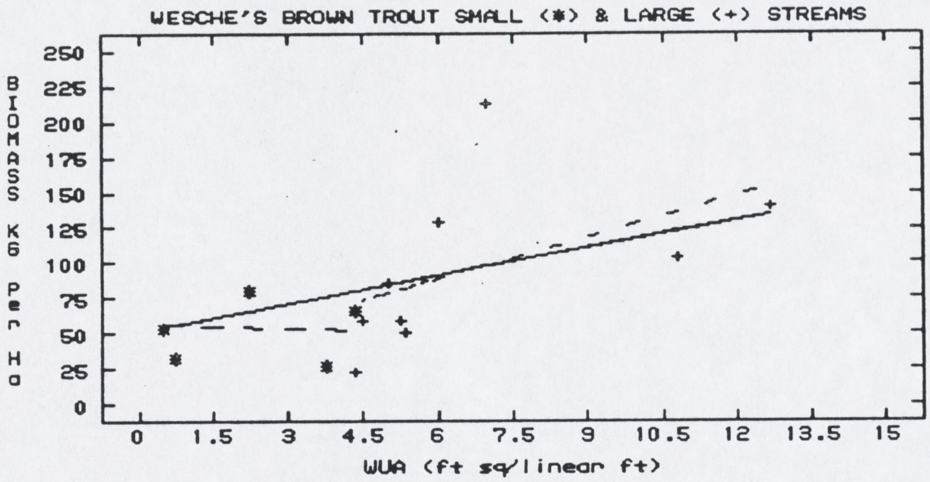
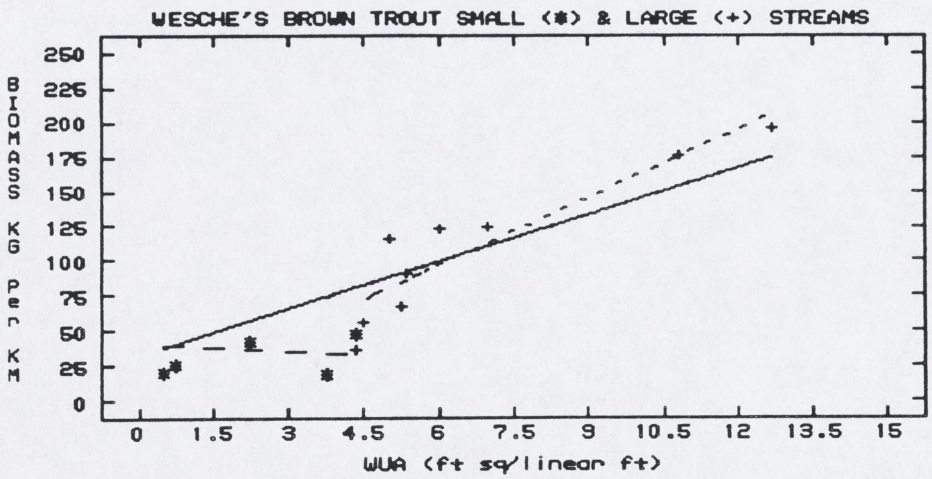


Figure 6-13b. Relationship between rainbow and brown trout biomass and Weighted Usable Area (Nelson 1980). For rainbow trout, there appears to be a fairly strong correlation, but the one for brown trout is probably not significantly different from 0.

Figure 6-14. Data set showing relationship between Weighted Usable Area (from IFIM) and Brown trout biomass from Wyoming streams. Data plotted from Wesche 1980. The upper figure is WUA versus biomass expressed per unit area. In the lower figure, biomass is expressed per length of stream. Wesche divided his data into those from small (average daily flow <100 cfs) and large (ADF >100 cfs) streams. The solid lines are least squares linear regressions through the combined data, the dotted line is through the data from large streams, and the dashed line is through the data from small streams. Note that there is no relationship between biomass and WUA in the small streams. The relationship begins to appear at WUA's .4.5 $ft^2 * ft^{-1}$ (large streams) and is quite strong ($r^2 = 0.83$) when large stream biomass is regressed on WUA.





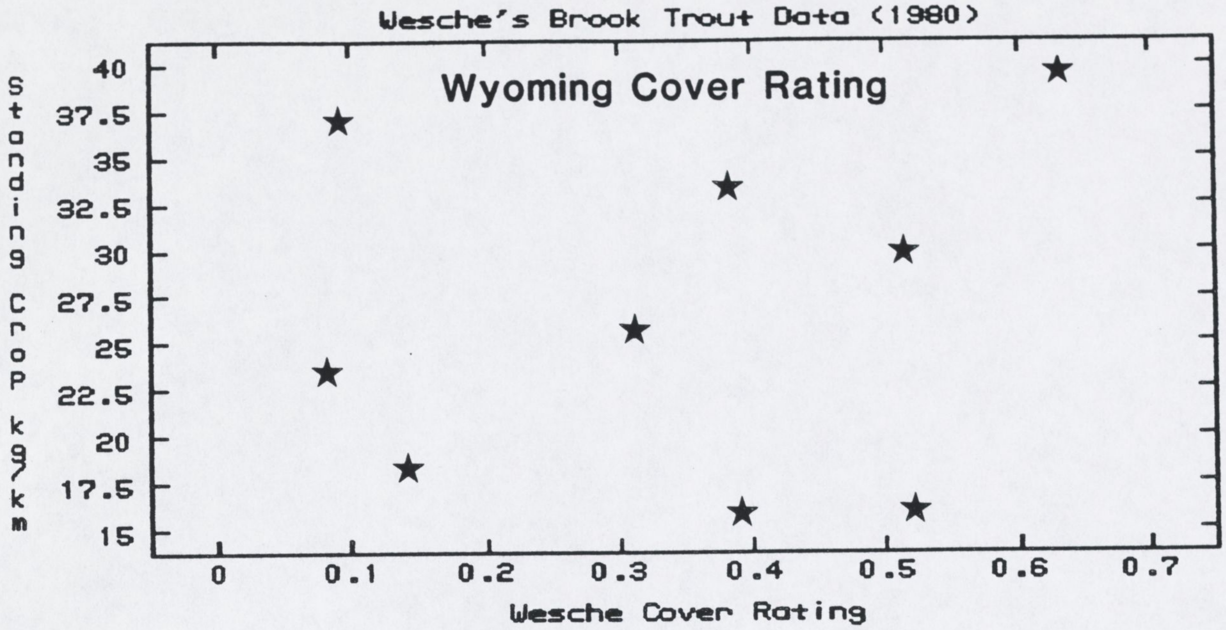


Figure 6-15a. Plot of brook trout standing crop versus the WRRI cover rating (Wesche 1980).

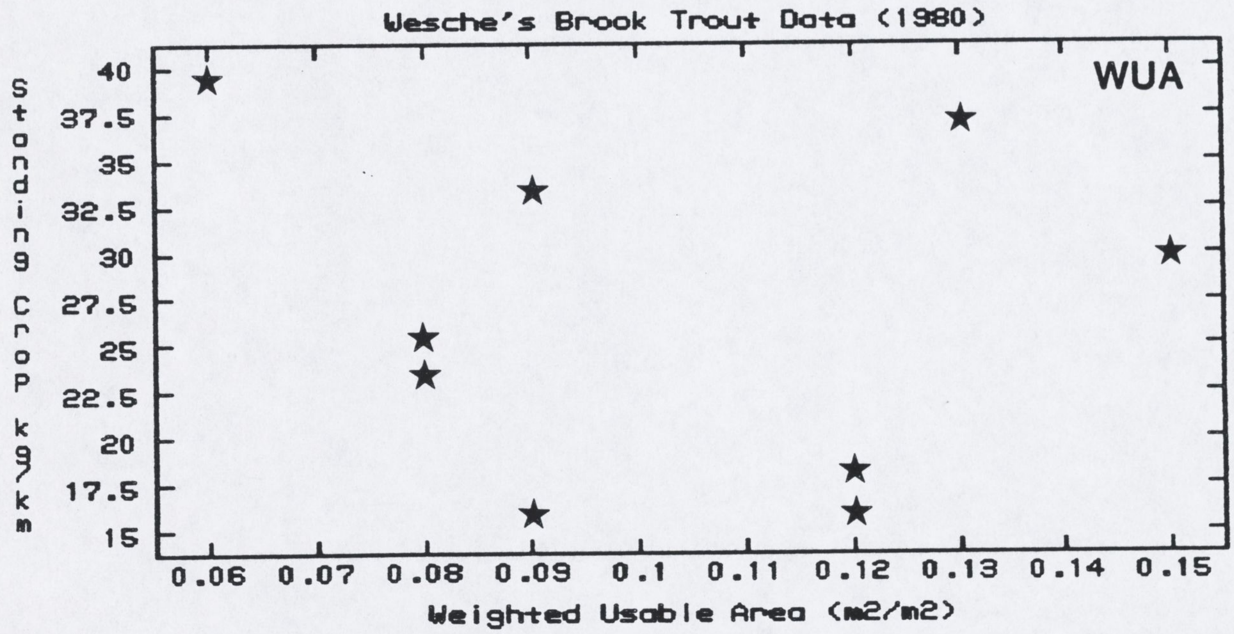


Figure 6-15b. Plot of brook trout standing crop versus biomass for brook trout (Wesche 1980)

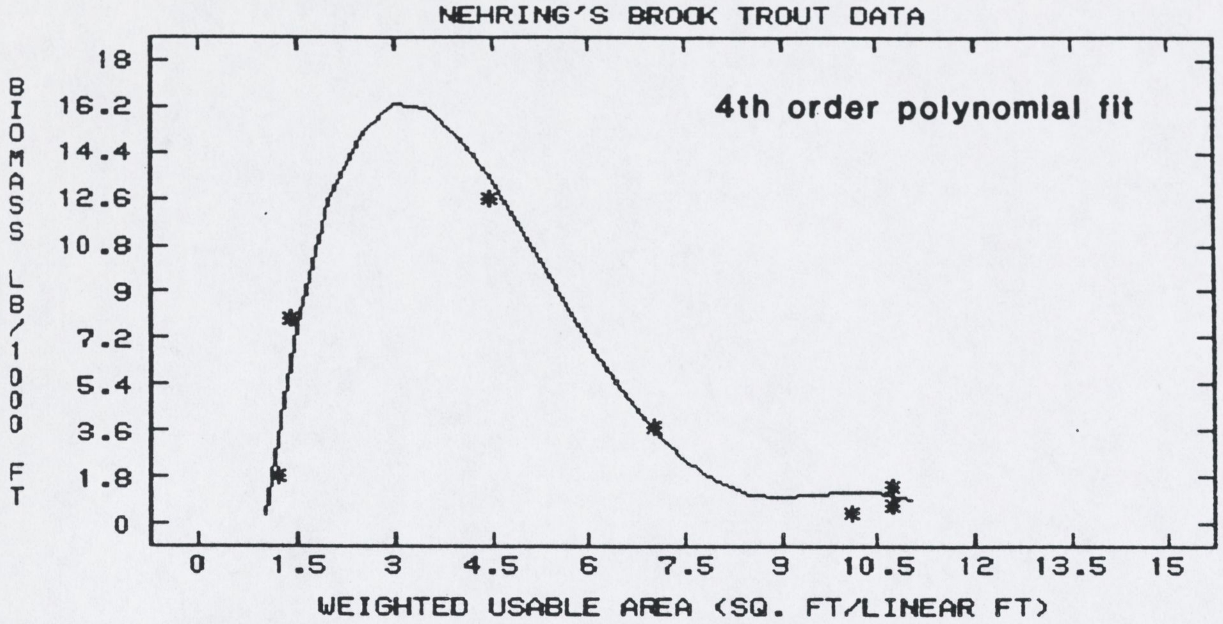


Figure 6-16a. Nehring's (1980) data on the relationship between brook trout biomass and WUA at Average Annual Flow. We fitted a 4th order polynomial to the data.

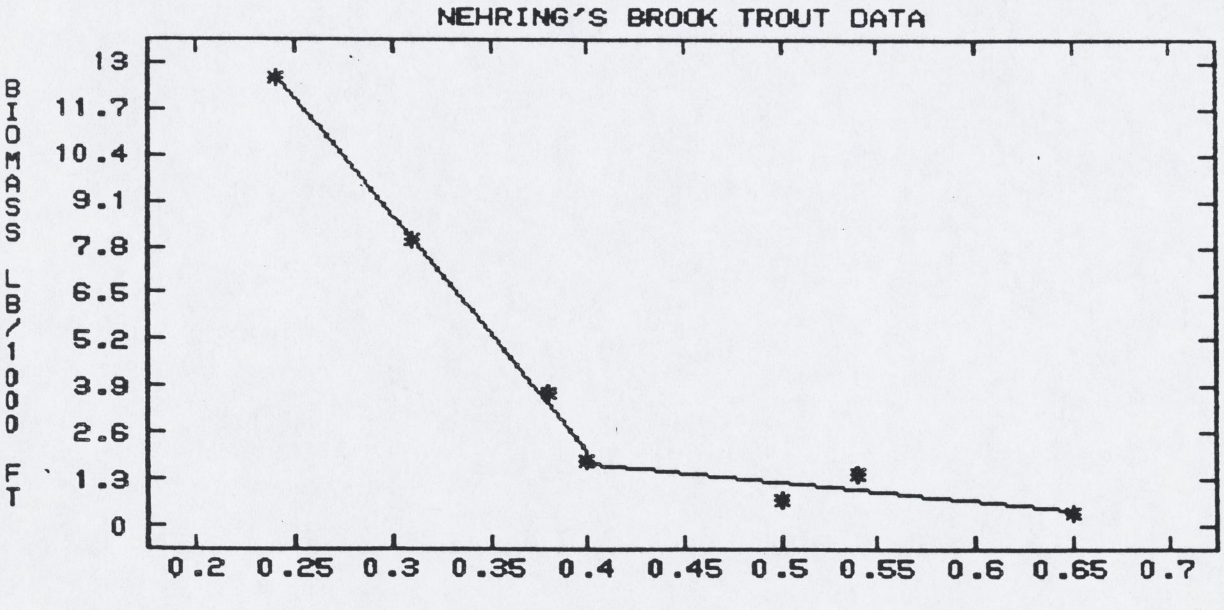


Figure 6-16b. Nehring's (1980) relationship between brook trout standing crop and depth.

there was a strong inverse linear relationship between brook trout standing crop and depth (Figure 6-16b).

Nehring also collected data on brown trout and found a positive strong relationship between stream width and brown trout standing crop (Figure 6-17a) and almost as strong a relationship between brown trout standing crop and WUA at Average Annual Flows (Figure 6-17b).

Kevern and Gowan (1984) calculated WUA for brown trout in a stream in central Michigan for two weeks prior to making fish population estimates in 1983 and 1984. Their results show no correlation in 1983 and a strong correlation in 1984 (Figure 6-18). They subsequently adjusted their 1983 data to include a lag period and to include biomass of brook trout and report a better correlation, but did not present the data. get.

Loar et al. (1985) conducted a very intensive and complex study to evaluate, among other things, the value of the USFWS HABTAT model coupled with the USFWS IFG4 model for some southern Appalachian trout streams, and the USFS WSP model for others, as a predictor of trout standing crop. Their 300 page report delves into most aspects of the HABTAT model and concludes with the observation that some variants of the WUA index were correlated with standing crop of brown and rainbow trout and others were not. For example, plots of standing crop (kg/km) versus adult WUA for rainbow and brown trout at the flows present when the standing crop was measured, showed essentially no relationship. They then explored a correlation matrix of abundance of and biomass of various sizes of fish against both existing WUA and PUA (WUA as a percent of total area). For brown trout, only 5 of the 40 possible correlations between abundance or biomass and WUA were significant and positive and another 5 were significant and negative. All but one of the positive correlations was with spawning area WUA, and the remaining one, adult (age 2+) abundance versus adult WUA explained only 28% of the variability. For Percent Usable Area on the other hand, 23 out of the 40 possible correlations were positive and significant. The strongest correlations were between spawning habitat PUA and abundance ($r^2 = 0.74$), while about 48% of the variability in adult (age 2+) abundance was explained by the PUA. For rainbow trout the situation was much worse with none of the possible correlations being significant.

To explore the hypothesis that the minimum monthly annual WUA or PUA rather than the WUA measured at the time of standing stock sampling was the critical value, Loar et al. (1985) made similar correlation matrices for abundance and biomass versus minimum monthly WUA and PUA. The results of these correlations were generally better. Eleven of the 40 possible correlations between brown trout abundance and biomass and minimum WUA were positive and significant, the strongest being age 2+ biomass versus adult WUA ($r^2 = 0.74$), total biomass versus adult WUA ($r^2 = 0.71$),

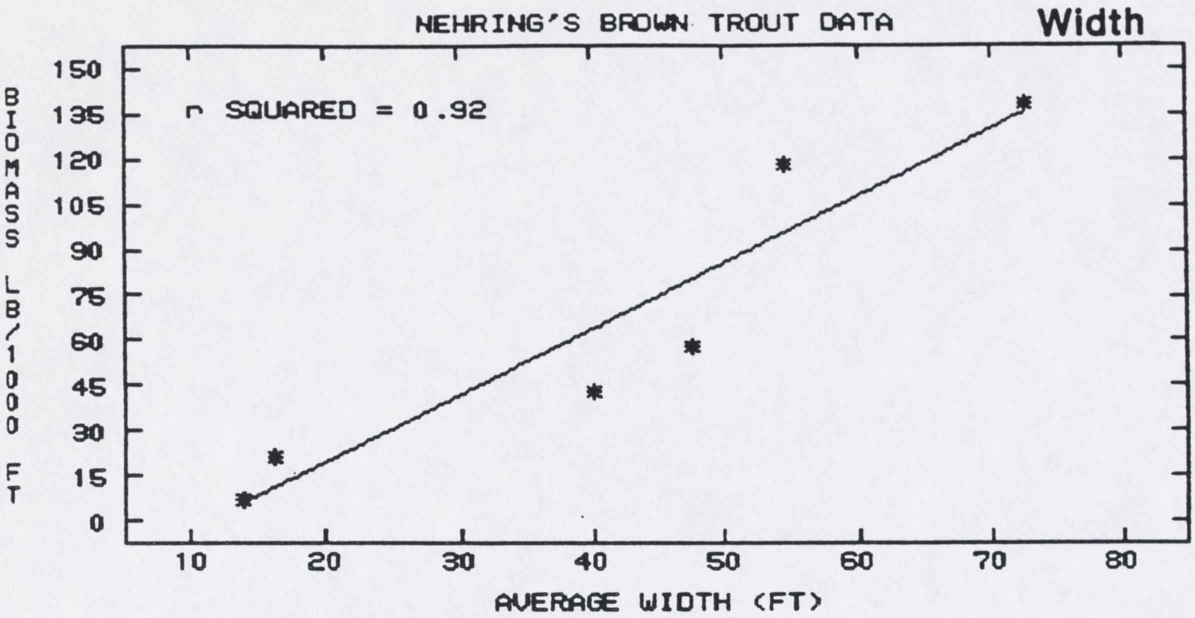


Figure 6-17a. Nehring's (1980) relationship between brown trout standing crop and stream width.

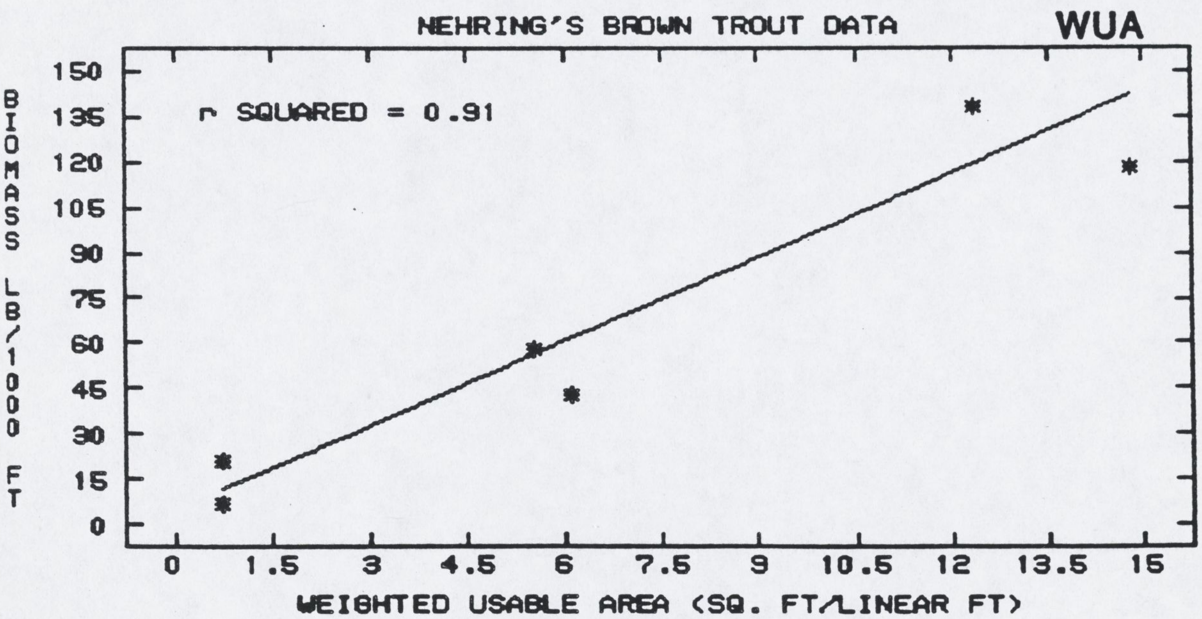


Figure 6-17b. Nehring's (1980) relationship between brown trout standing crop and WUA at Average Annual Flow

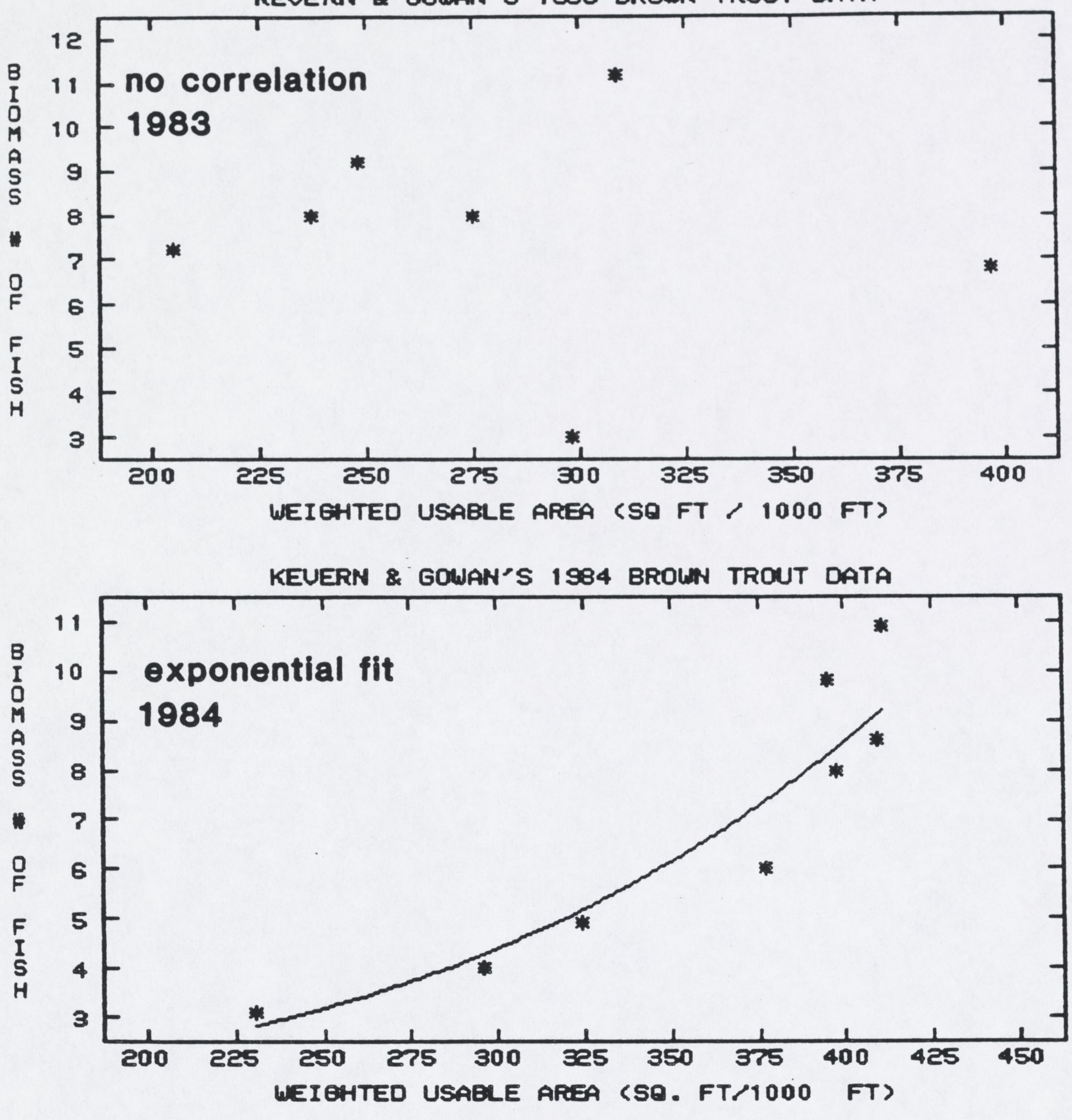


Figure 6-18. Kevern and Gowan's plot of biomass of brown trout (# of fish) versus average WUA over the preceeding two weeks for two different years.

total abundance versus spawning WUA ($r^2 = 0.58$) and adult (age 2+) abundance versus adult WUA ($r^2 = 0.53$). Interestingly, adult and total biomass and adult abundance were equally strongly negatively correlated with fry WUA. For correlations with minimum PUA, more (29 out of 40) were correlated with abundance and biomass, but none as strongly as the best of the WUA correlations.

For rainbow trout, none of the correlations relating minimum adult, juvenile, and spawning WUA and PUA to adult and juvenile abundance and biomass were significant, but there were a few strong correlations between incubation and fry WUA and PUA and both abundance of all life stages and age 0 biomass. The strongest of these were between incubation minimum WUA and age 0 biomass ($r^2 = 0.74$), fry minimum WUA and age 0 biomass ($r^2 = 0.67$ and incubation with and total biomass ($r^2 = 0.62$).

We calculated the average abundance of adult (age 2+) fish sampled in each stream (from tables in the Loar et al. 1985 report) and plotted them against the mean annual WUA, the minimum annual WUA, and the minimum % WUA for both brown and rainbow trout (Figures 6-19 and 6-20). The resulting plots show essentially no correlations for the adult rainbow trout. For brown trout, 67% of the variation in adult abundance is explained by the mean annual WUA, and 46% by the minimum annual WUA.

Nelson et al. (1984) did the first validation study we have seen relating standing crop of non-salmonids to WUA. Both adult numbers and adult biomass were strongly correlated with both adult WUA and juvenile WUA occurring at the median discharge for 30 days prior to population sampling (Figure 6-21), but juvenile numbers and biomass were correlated with neither.

request

We find all these results taken together to be difficult to describe under a unifying theory. The real question here is whether fish production and standing crop would be decreased if the discharge were changed in such a way that the Weighted Usable Area decreased. None of these validation studies has attempted to explore that question. Nearly all of them were done in undiverted streams and none of them exposed fish to altered flow regimes. The large variability of which version of WUA is correlated with which age of fish does not increase our confidence in choosing a particular form of the WUA output for use in attempting to arrive at realistic and appropriate instream flows.

Must manipulate X and observe change in Y.

VALIDITY OF MODELS USING COVER DATA

The only model we considered in this category is Wesche's 1980 WRRI cover rating method. Wesche's results are shown in Figure 6-22. The strength of the correlations suggests that a cover term should be included in any predictive salmonid standing crop

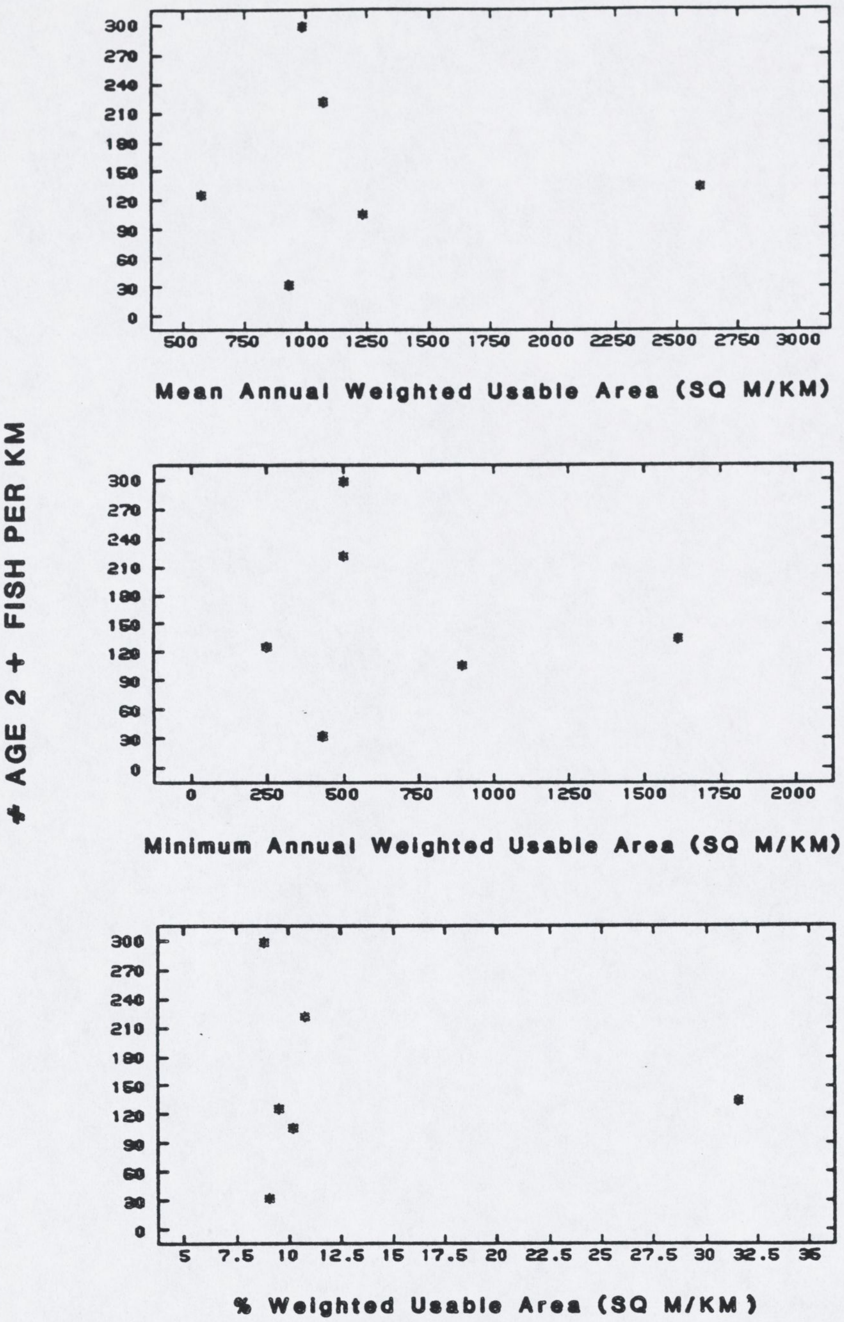


Figure 6-19. Adult rainbow trout Weighted Usable Area and standing crop data from Loar et al. 1985. Numbers of adults = averages from all (4 or 5) sample periods (Table 4-38). WUA and %UA from Tables 4-23 and 4-25. Minimum WUA from Figure H-5. Table and figure numbers are from their report.

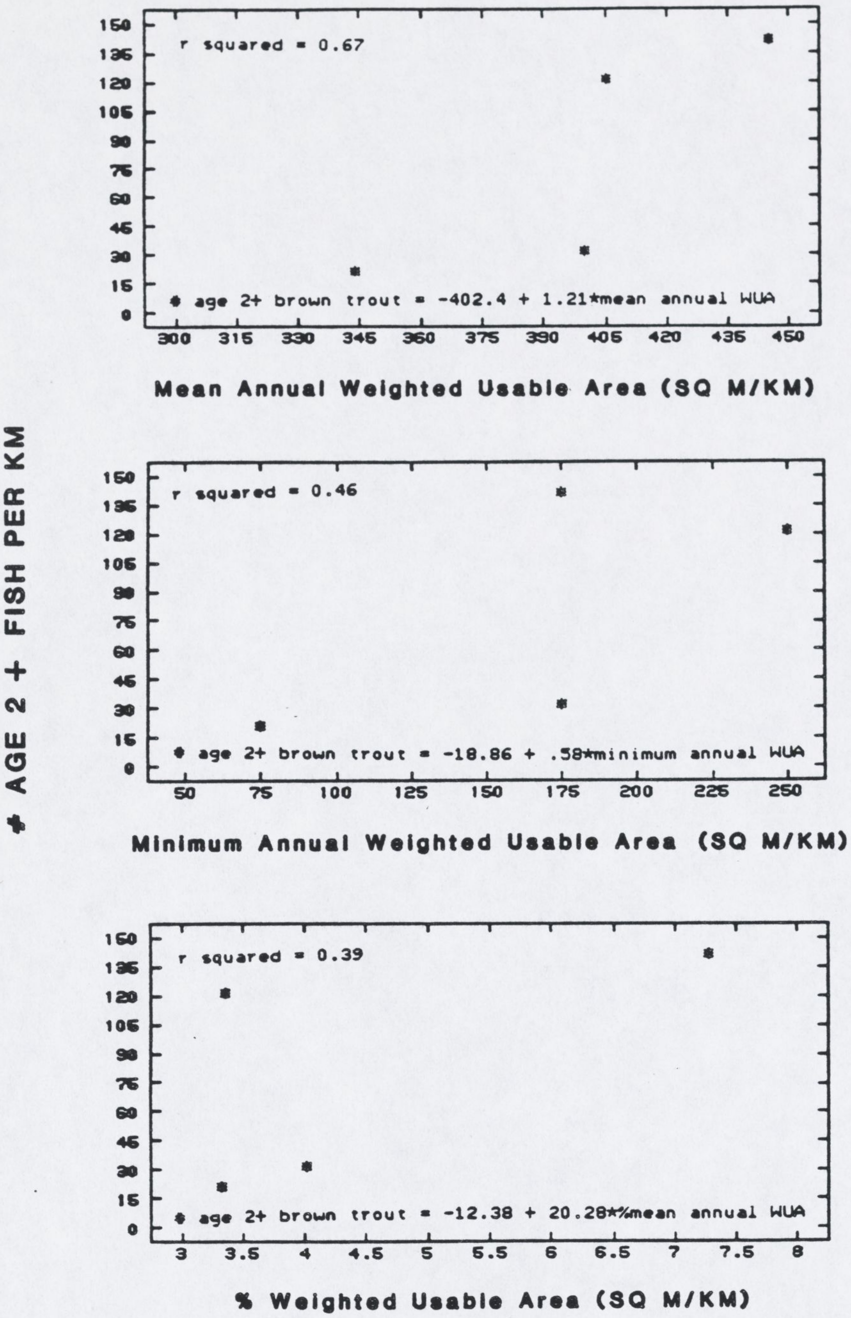


Figure 6-20. Adult brown trout Weighted Usable Area and standing crop data from Loar et al. 1985. Number of adults = averages from all sample periods (4 or 5) from their Table 4-34. WUA and % UA from Tables 4-22 and 4-24. Minimum WUA from their Figure H-10.

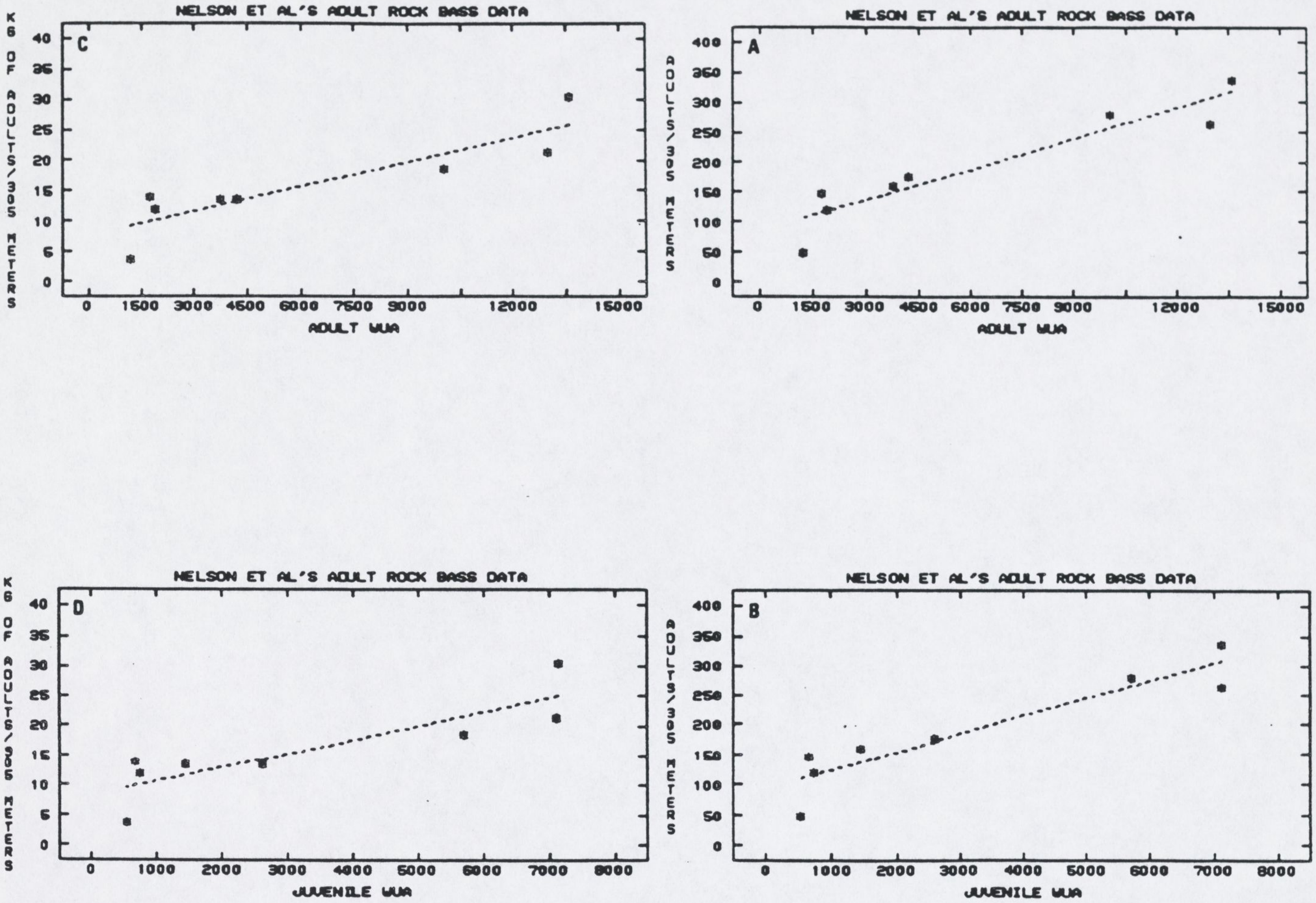
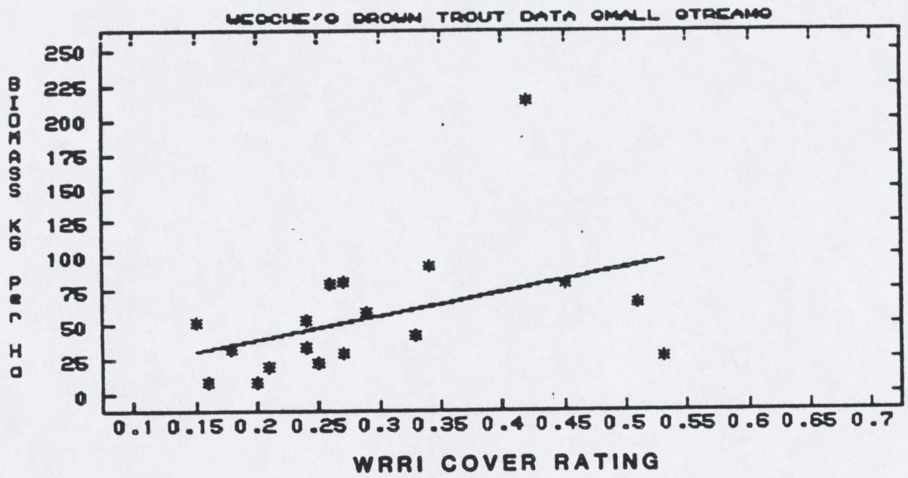
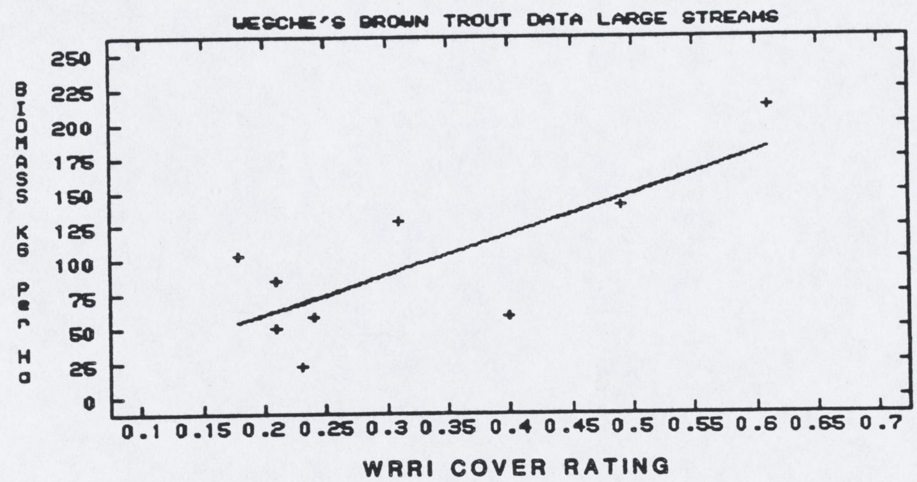


Figure 6-21. Nelson et al.'s (1984) regressions of adult rock bass numbers and biomass on adult and juvenile WUA per 305 meters of stream. Note that correlations are similar for both adult and juvenile WUA. Juvenile numbers and biomass were not significantly correlated with either adult or juvenile WUA.

Figure 6-22. Correlations between an index of cover (the WRRI cover rating, Wesche 1980) and brown trout biomass. The correlation is fairly strong (r = 0.76) for large (ADF >100 cfs) streams, but is weak (r = 0.42) for small (ADF <100 cfs) streams. (If the Little Laramie River #2 (ADF = 103 cfs) data point is placed in the small data set as Wesche did, the r = 0.61, somewhat better)





model, but the degree of scatter suggests that this model alone would be insufficient to predict standing crop.

VALIDITY OF FWS HSI MULTIVARIATE MODELS

request

Gilbert (unpublished) estimated standing crop of seven warmwater fish species in Georgia and plotted there against Habitat Suitability Index values determined using the USFWS HSI models for the species in question. He obtained very poor correlations, all illustrated in Figure 6-23 and fit by us with the best (if any) regression line.

Li et al. (unpublished) generated some of their own SI curves based on standing crop data and used the published curves for others, and tested 3 types of data aggregation in an HSI model for cutthroat trout and coho salmon in Oregon. The 3 techniques used were geometric means of the Suitability Indices, multiplicative total of the Suitability Indices, and lowest value of the Suitability Indices. For the coho salmon models, the correlation coefficients between predicted and actual standing crops were -0.19, -0.21, and -0.07. For cutthroat trout they were -0.30, -0.33, and -0.28 except when corrected for competitive effects when they were -0.11, -0.32, and -0.09. In other words, the models explained almost none of the variation in standing crop and that they did explain was in the direction opposite from expected.

request

Trial et al. (unpublished) tested HSI models for blacknose dace, common shiner, fallfish, and Atlantic salmon and brook trout. They published sufficient information to determine the correlation coefficients for the predictions of numbers of fish per hectare for 3 species (Figure 6-24). Correlations were very poor except for Atlantic salmon, which, however, was determined from 3 data points spaced in such a way that a strong correlation could not have failed to result.

VALIDITY OF REGRESSION MODELS

Layher (1983) constructed his own HSI models using empirical HSI curves and linear regression (described in Methodology Summary Forms) based on data from Oklahoma. He then attempted to validate them by applying the models to data collected in Kansas streams and discovered that different variables were correlated with standing crop in Kansas than in Oklahoma. Thus, the models were invalid when transported to Kansas. They may also be invalid in Oklahoma when used in a predictive sense. Layher did not test them against a new data set.

Binns and Eisermann (1979) also created an HSI-like model using linear regression (see Methodology Summary Form) and subsequently tested their model on 8 additional Wyoming streams. The R^2 between standing crop and their estimate of it for the test

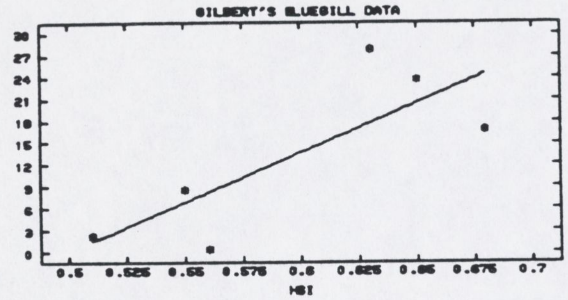
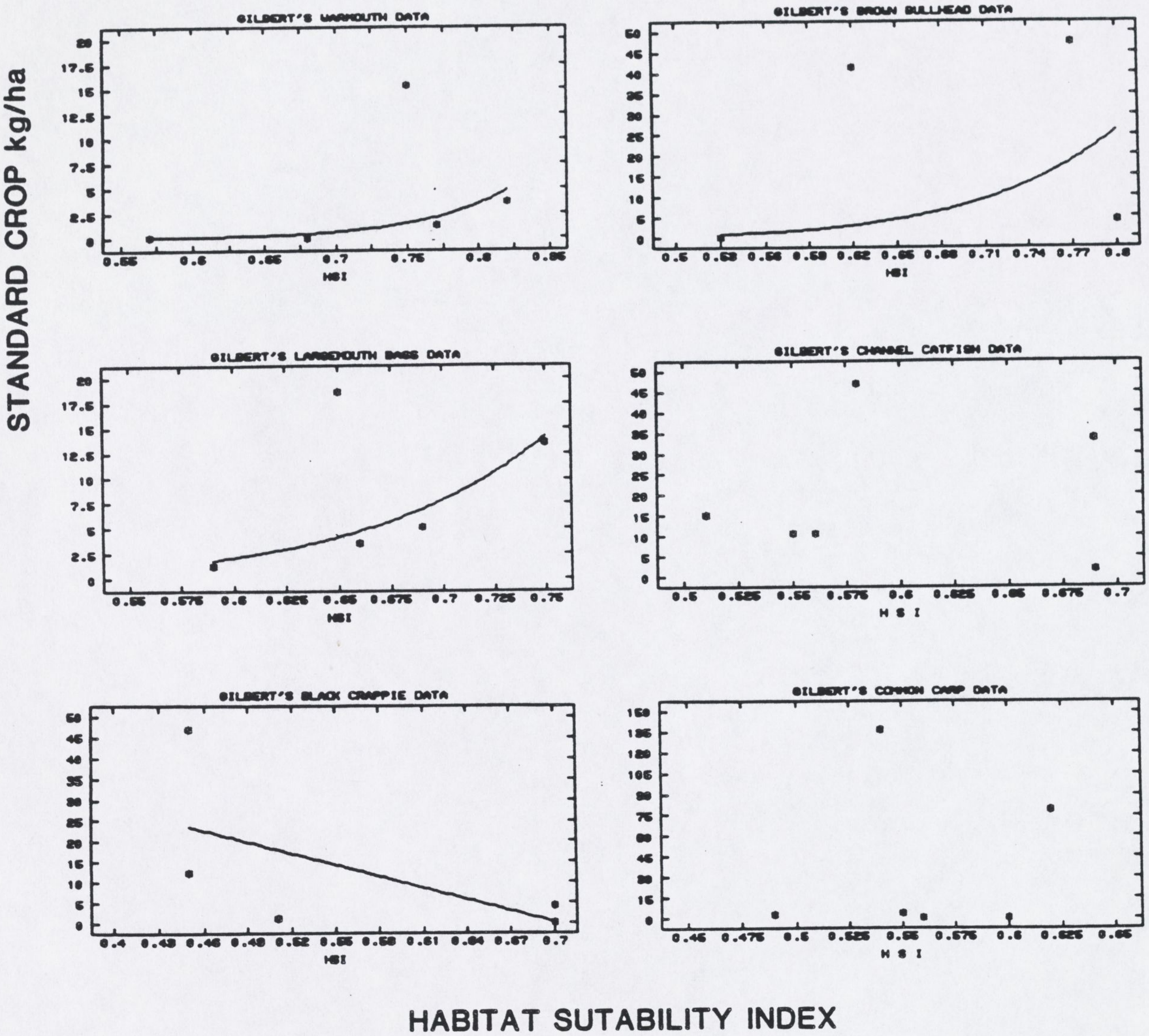


Figure 6-23. Plots of Gilbert's (unpublished) Habitat Suitability Index data against standing crop. The regression lines were fit by us using the most highly correlated of linear, e^y, Log, or x^n transformed data.



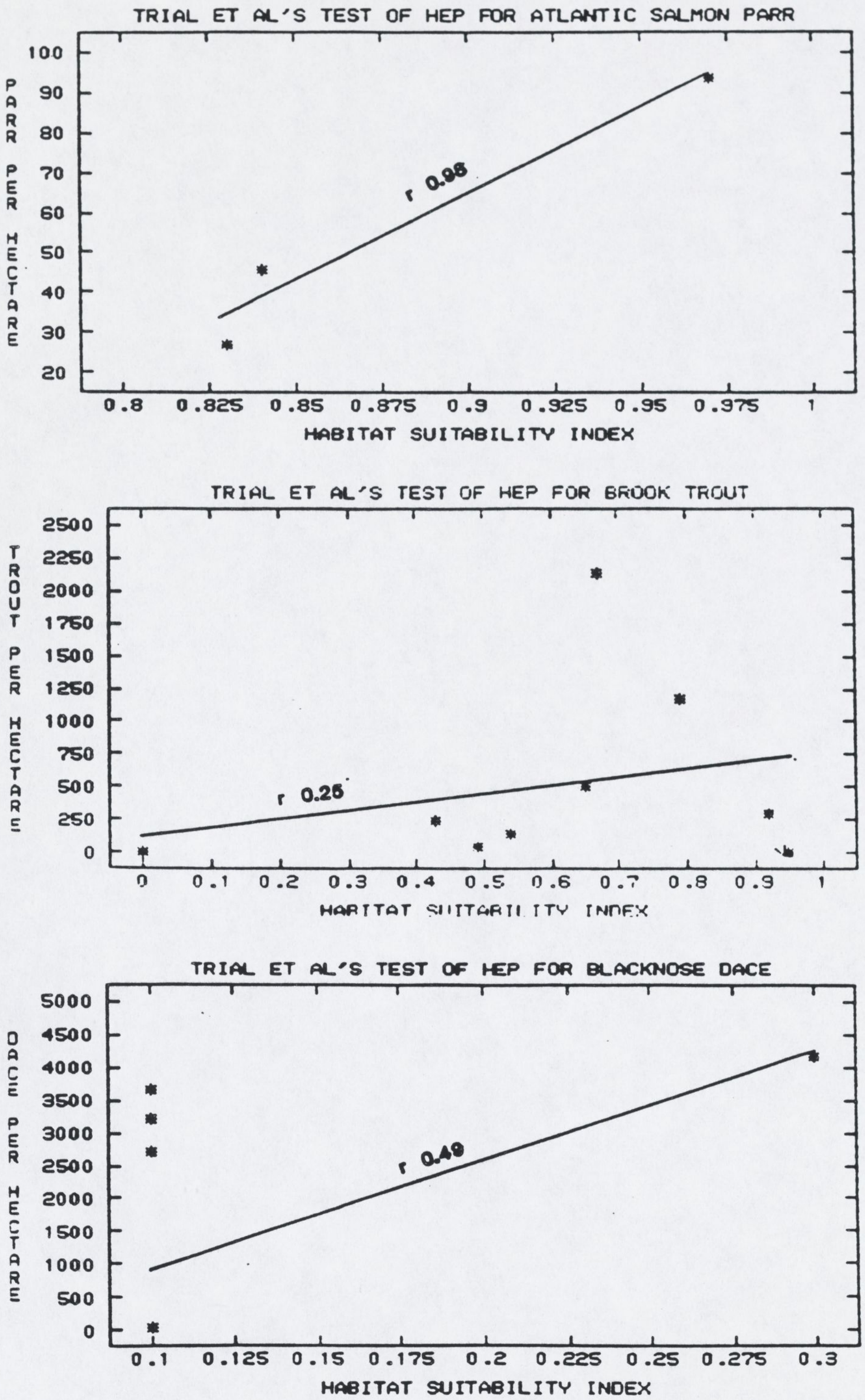


Figure 6-24. Trial et al.'s (unpublished) test of three HSI models. Note that because of the spacing of the data on the Atlantic salmon model, a strong correlation is inevitable.

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streams was 0.965. By far the strongest validations we have seen for any predictive model.

Annear and Conder (1983) applied Binns and Eisermann's Model II to ten additional Wyoming streams at low summer flows and found strong correlations ($r^2 = 0.83$) between measured and predicted HQI, further supporting the use of this model in Wyoming.

The other regression models reviewed were not tested against additional data sets, so their validity cannot be assessed.

CHAPTER 7: MAKING RECOMMENDATIONS OF APPROPRIATE INSTREAM FLOWS

[This chapter was not finished in time for this draft, but will be available for the April 30 meeting: It will describe the various ways that have been used to arrive at an instream flow recommendation from the response variables produced by the various models].

CHAPTER 8: CONCLUSIONS

This chapter is also incomplete, but in it we will draw conclusions about the state of the art of instream flow modeling and the significance that can be attached to results from any particular model. Our conclusions are not yet fully formed and will be influenced by input at the review meeting, but are likely to include these points:

- There is no concensus regarding which input variables are important or on how many to use for any species. Until that is sorted out, unvalidated models should be regarded as hypotheses.

- The choice of unmeasurable habitat quality indices such as WUA, HSI, and HQI as response variables (rather than measurable variables such as standing crop) puts a difficult burden of proof on the model builder, but one which should be required prior to dissemination and acceptance of the model.

- There are very few data at present supporting the validity of any instream flow or habitat quality model for predicting the resulting fish population or other effects of partial diversion. Almost all existing tests of these models have been made on undiverted streams.

- There is not much evidence that a completely transportable model is likely to be developed. There is increasing evidence that geographically specific calibration or weighting will be required for most predictive models.

- Empirical regression models, although probably usually specific to limited geographical areas and stream types, show promise of being highly predictive.

- Hybrid models, such as the IFIM, which uses site-specific data transformation coupled with an arbitrary mathematical structure and no parameterization, are descriptive for some species and life stages, but not for others. No pattern of descriptiveness is clear as yet, and no predictive studies using this model have appeared.

- Top-down conceptual models using weighting curves from the literature (HEP models) have not been shown to have any descriptive ability, much less any predictivness.

There is a major need for studies that specifically examine the effects of diversions, and for modeling efforts that go through the proper cycle of development, experimental testing, and revision until the resulting models are both descriptive and predictive. *

CRITIQUE OF INSTREAM FLOW METHODOLOGIES

Robert J. Behnke

May, 1986

ABSTRACT

During the past 30 years numerous methodologies have been developed to assess instream flow needs of fishes. A basic problem is that no methodology is likely to have success, on a broad scale, to accurately predict changes in abundance or biomass of a species with changes in flow. This is due to limitations for making predictions based on variable biological systems and the failure of any model to accurately take into account all of the subtle interacting factors that determine the well-being of a species in a particular environment in addition to physical habitat limitations. The IFIM of the US Fish and Wildlife Service is a widely used standard model that offers the advantage of comparing habitat changes (expressed as weighted useable area or WUA) for different life history stages of a species throughout an annual cycle. The problem with WUA, however, is: into what biologically meaningful terms can it be translated? It cannot accurately predict changes in numbers or biomass because the IFIM model is faced with the same problems that limit any predictive habitat model.

INTRODUCTION

It has long been the goal for environmental assessment and prediction methodologies to accurately predict quantitative changes in target species as the consequence of environmental changes. For fishes living in rivers to be subjected to a new flow regime the logical assumption has been that a new flow regime will effect changes in fish habitat which, in turn, will effect changes in the population of the target species. In the past, most efforts have been aimed at the determination of minimum flow standards. The assumption is that if flows fall below a designated minimum, an unacceptable decline in the population of the target species (or groups of species) will result. The problem has been that any predicted increase or decrease in the fish population invoking a direct cause-and-effect relationship between flow-habitat-fish has not been quantitatively verified. Quantitative changes in a fish population can not be accurately predicted from changes in the flow regime. Proponents of water projects that propose to change the flow regime in a river typically have precise figures on the value of the water; for example, the value of electrical generating capacity expressed as generation of electricity per cubic-foot-per-second flow. To meet a recommended flow standard, the costs incurred for lost generation capacity can be quantified. Water development proponents demand that the benefits to the fish be similarly quantified from a recommended flow, and this can not be done with any precision.

During the past 30 years, a variety of techniques have been used by state and federal agencies to make flow recommendations. None have been able to demonstrate their ability to quantify changes in aquatic values with changes in flow. During the past 10 years, the U.S. Fish and Wildlife Service has developed a standardized "Instream Flow Incremental Methodology" (IFIM), which allows a habitat model for different life history stages of a species to couple with a hydraulic model to quantify changes in habitat (expressed as weighted useable area [WUA]) with changes in flow by computer simulation. The problem that has become apparent in recent years is that too many people were captivated by the "illusion of technique". They had a naive faith that confused objectivity and quantification with biological reality. Although IFIM can be useful to provide insight into certain limiting factors such as spawning and incubation flows and flows with velocities excessive for the well-being of newly hatched fish, the fact remains that changes in WUA do not provide a basis to accurately predict changes in abundance or biomass of the target species. Changes in flow can be precisely translated into changes in WUA for a target species but WUA can not be accurately translated into changes in numbers or biomass of the target species. The failure to accurately predict changes in a fish population with changes in flow is not so much the failure of the IFIM methodology or any other methodology but rather the limitations for any predictions imposed by natural variation. An understanding of the limitations on prediction (or prophesizing the future) has long been a basic tenet in philosophy and logic and can be roughly expressed as follows: Accurate predictions based on observations (or data) from the present and past are possible only if the system under observation is stable, isolated and highly recurrent -- and such systems are extremely rare in nature. For example, long term and accurate observation and data collection on tidal fluctuations at a point on a seashore would allow accurate predictions of future tides (a tide table) because tides are governed by the constancy of the law of gravity and the solar system (but even with such a stable and recurrent system, unpredictable wind events can alter the accuracy of any predication).

Natural, uncontrollable variation of biological systems such as fish communities in rivers impose severe limitations on any predictive model. Hall and Knight (1981) produced a compendium of documentation on natural variation of populations of salmonid fishes in streams which clearly emphasizes this point. An understanding of the niche concept of a species will also make clear the limitations for accurate predictions of population change associated with any suspected cause. A species "niche" is the total interaction of a species with the biotic and abiotic components of its environment. The current Hutchinsonian niche concept, widely applied in ecology, conceives the niche to be "n" dimensional (unlimited number of factors influencing well-being). The "basic" or fundamental niche and "realized" niche of a species are of different "volume". That is, environmental components such as temperature, living space, predators and competitor species interact to restrict the abundance of a species in a particular environment. Because of this, the basic "niche" is reduced to the realized niche (and the population of the species exists at a lower than maximum level -- this distinction between basic and realized niche relates to the distinction between "carrying capacity" and "standing crop" or biomass of a species and concerns problems of translating WUA (weighted useable area) into biologically meaningful terms to be discussed later). The changes involved in determination of a species realized niche, introduces the concept of "niche shifts". Niche shifts may occur when two or more species interact in such a way to partition the environment and reduce interspecific competition which allows for their coexistence. When niche shifts occur, "preferences" or "suitabilities" of different environmental factors such as depth and velocity can be expected to change.

An understanding of niche theory with its "n" dimensions and "volume" subjected to continual change makes clear that any habitat model based on very few dimensions of the niche (such as depth and velocity) and expressing these dimensions as a static, deterministic, two-dimensional "suitability index" is under severe constraints for accurate prediction of niche changes expected (the new realized niche) from a change in a flow regime, especially if attempts are made to express the predicted changes in terms of abundance and biomass. Such models can be expected to work best for species with a very narrow niche, where complete dependence of the species well-being can be related to a single environmental component, such as might be conceived for a rare species of fish that is only known to occur in beds of watercress, or koala bears known to live only in eucalyptus trees. Such species, however, are rare.

The limitations of ecological models to correctly predict future population changes associated with environmental changes was clearly recognized by one of the early promoters of the use of computer simulation models for environmental assessment (Hollings 1978). Hollings emphasized that the best models can only be a highly condensed abstract of nature, that accurate predictions should not be expected, and to expect the unexpected.

HISTORICAL REVIEW

In former times dams were constructed and rivers regulated in accordance with the purpose of the dam without regard to fish. In those rivers with valuable fisheries some minimum flow was recognized to be essential. Various formulas for minimum flow were used such as the lowest natural flow for seven consecutive days during a 10 year period. The only biological basis for such minimum flows was the assumption that the present fish fauna of the river had survived such low flows in the past (at least for seven days) and they could survive at such flows in the future. In the 1950's the public became more environmentally aware and fisheries values became better documented, especially for anadromous salmon on the Pacific Coast. Many studies were initiated, with mixed success, to attempt to correlate annual or seasonal flow regimes in a specific river or a group of rivers in a geographic area with salmon production.

The earliest attempts at developing quantitative methods to relate flow recommendations to fish habitat concern the transect method whereby a transect is placed across a stream channel to measure depth, velocity and wetted perimeter (area of channel covered by water at different flows). Typically, a section of a stream designated as a "critical riffle" would be selected for the transect measurements. Arbitrary values would be selected for depth, velocity and/or wetted perimeter (for example, six inch depth, one foot-per-second velocity, and/or 70% wetted perimeter) at the critical riffle site which would be achieved with the minimum flow recommendation. Unless a series of transects are made at varying flows to derive empirical data, a formula (Manning's formula) is needed to predict the flow which meets the required depth and velocity. Elements of Manning's formula such as "roughness coefficient" and "slope" make prediction prone to considerable error. Nehring (1979) compared the "R-2" cross section method and the IFG4 method (used with IFIM studies) to check for the error between predicted and actual velocities. For the R-2 cross method 30% of 97 predictions were within 10% of the actual velocities and 7% were in error by 100-500%. For the IFG4 method, 35% of the predicted velocities were within 10% of the actual velocities and 4% were more than 100% in error. Another criticism of the simple transect method is that flow recommendations derived from it can not be readily related to biological reality. That is, if the actual flow falls below the recommended flow by 10% or 20%, or is maintained above the recommended flow by a known amount, how is the fish population affected? What parameters change? by how much? Such questions can not be answered with any confidence.

The assumption inherent in the simple transect method (its methodology) is that if a flow meets certain criteria for depth, velocity and wetted perimeter at the "critical" site, then the stream channel in other sections of the stream will contain sufficient water to maintain certain desirable habitat features. This assumption was given some credibility by Wesche (1973) who found that when flows decreased below about 25% of the average annual flow (=average daily flow), optimum trout habitat under the streambanks was rapidly lost due to declining water levels. Nehring (1979) obtained 18 flow recommendations with the R-2 cross section method which ranged from 15 to 44% of the average daily flow for the streams studied with an overall average of 26.4% of the average flow based on depth and velocity measurements made at multiple transects.

Thus, for a low effort method where significant conflicts over flow recommendations are not anticipated, a transect technique performed by an experienced and knowledgeable biologist would be acceptable and the flow recommendation in relation to the well-being of the target fish species could be expected to "be in the ballpark" but not quantitatively predictive (but no other technique or methodology can claim better predictive power).

For situations where much is at stake in regards to proposed flow changes, it was realized that more sophisticated and defensible techniques were needed (to produce evidence that might better hold up in court). In 1960, the California Department of Fish and Game developed a methodology to assess flows below dams. This methodology assumed that the basic requirements of fish include food, shelter and reproduction and that habitat parameters for food (food producing areas), shelter (resting areas with suitable cover), and reproduction (spawning areas) can be quantified and quantified changes in habitat quality can be related to flow changes. Fisheries biologists of the Pacific Gas and Electric Co. (PGE) further developed and refined the methods and methodology and adapted the model for computer simulation (Waters 1976). The PGE model is the direct antecedent of the present IFIM of the U.S. Fish and Wildlife Service.

Another well-known flow assessment technique is popularly known as the Montana or Tennant method (Tennant 1976). This method requires only USGS flow records for a stream and flow recommendations are based on percent of long term average daily flow for a stream. These range from 10% of ADF for "short term survival" to 60-100% of ADF for "optimum" fishery flows.

Allen Binns, Wyoming Game and Fish Department, developed a habitat assessment methodology for Wyoming trout streams with a quantitative output expressed as the Habitat Quality Index (HQI) (Binns and Eiserman 1979). Binns' HQI is the only widely known model that directly relates habitat variables to fish biomass (only trout). Only two flow parameters -- late summer base flow as percent of ADF, and difference between maximum and minimum flows (least difference = best and greatest difference = worst habitat conditions) -- are included in Binns' habitat model. Techniques would have to be developed to relate changes in the other habitat parameters to changes in flow in a consistent manner before the Binns HQI model could be utilized to predict changes in trout biomass to changes in flow. If attempted, a significant element of subjectivity would be introduced to the habitat assessment and would require a considerable amount of experience and expertise on the part of the biologist to make it work. Thus, I suspect that Dr. Binns might be able to predict changes in trout biomass in Wyoming streams from changes in flow with a fair degree of accuracy (ca. ± 50%), but other biologists in other areas could not duplicate his results.

A comprehensive review of instream flow methods and methodologies was prepared by Wesche and Rechard (1980). For the remainder of my critique I will mainly concentrate on the IFIM of the U.S. Fish and Wildlife Service because this has become the "standard" and often the required method of federal and state agencies. It is important that parties involved in flow determination decisions understand certain concepts and limitations to avoid expensive and meaningless work or misdirected application of a method where it does not apply such as attempting to use WUA for mitigation trade-offs. For example, a stream section to be inundated by a reservoir can be calculated to have a quantified amount of WUA for brown trout and rainbow trout and the project developer might offer to purchase or improve another stream with an equal amount of WUA (attempted by Denver Water Board for Two Forks Project on South Platte River). What must be understood is that WUA values are not equal between different streams (when correlated with fish biomass) and are not interchangeable.

Also, one overwhelming factor may preclude the use of habitat to predict occurrence and well-being of a species. If empirical evidence demonstrates that species A is never found in the presence of species B, then the presence of species B will exclude species A, no matter what a habitat model predicts for the success of species A. Such a situation occurs in the Salt River drainage where the presence of smallmouth bass excludes the occurrence of many native species such as the spike dace, loach minnow, and Gila chub.

My critique is not intended to be a negative criticism of IFIM. I believe most who have been intimately involved with IFIM will agree with the theme of my critique, but perhaps not all of the details. My words of caution are intended for those involved in negotiations and discussions of impact analysis with administrators of state and federal agencies who may have only a rudimentary and naive understanding of a particular situation causing them to invoke a reflect response demanding "baseline study", "IFIM study", etc. when such studies may be meaningless to resolve a particular problem.

DETAILS OF PROBLEMS FOR PREDICTION

The great advantage of IFIM over other methodologies is its ability to quantitatively display changes in WUA (assumed to represent the habitat quality of target species) with changes in flow, which can be plotted on an actual or proposed annual hydrograph. This allows negotiators to discuss trade-offs and mitigation for proposed projects in a quantitative manner. As such, IFIM was quickly embraced by federal agencies as a long-sought saviour to their problem of quantification of gains or losses to the biological system from flow changes. For many, the hard question of what does WUA relate to, was ignored or not even considered. When the question was asked and tested, the results were a disillusionment to many and a confirmation to those who were aware of the limitations of prediction discussed above.

The U.S. Fish and Wildlife Service contracted for validation studies of IFIM and its habitat suitability index curves (HSI), used for both IFIM and HEP (Habitat Evaluation Procedures). The Electric Power Research Institute (EPRI) contracted for its own evaluation of instream flow methodologies (Morhardt 1985), and the U.S. Department of Energy contracted with the Oak Ridge National Laboratory to test habitat evaluation models in southern Appalachian trout streams (Loar 1984). The results from attempting to correlate WUA or HSI values with fish numbers or biomass have not been encouraging. Numerous papers documenting the failure of HSI to correlate with abundance or biomass of the target species are contained in a volume edited by Terrell (1984). Layher and Maughan (1985) concluded that "broad niche" species can not be adequately represented by a simple suitability curve, that reliable habitat models may only be possible for small, homogeneous areas, and that HSI should be used only for planning and not decision making.

Several studies documented that habitat "suitability" or "preference" changes in a species in relation to daily and seasonal differences, the presence or absence of other species and other complex factors (Larimore and Garrels 1985, Li and Schreck 1984, Loar 1984, Sheppard and Johnson 1985, Moyle and Baltz 1985). That is, WUA computed on depth, velocity and cover for a species would vary when recorded at different times and/or in different streams with different fish communities. I became aware of this problem a few years ago when I was advising the U.S. Justice Department regarding a claim for instream flow on the Red River, New Mexico. The analysis showed WUA for rainbow trout was much greater than was the WUA for brown trout, yet brown trout were completely dominant over rainbow trout in the river. I realized that if the case went to court, new suitability index curves would have to be made specifically for the brown and rainbow trout of the Red River, which would agree with their relative populations in the stream. The U.S. Fish and Wildlife Service species habitat models now contain the recommendation that suitability curves should be based on site-specific studies. The problem here is that studies to obtain detailed data on a sufficient number of individuals of a species to compute original suitability curves for such parameters as depth, velocity, substrate, and cover, is time consuming and expensive.

A recent paper by Mathur et al. (1985) contains a strongly negative attack on IFIM (but it should be recognized that these authors had an "ax to grind"). In any event, it's obvious that the "bloom is off the rose" of IFIM (and HEP), but the critiques and criticisms of both the biological and mathematical-statistical bases of IFIM should have a salubrious effect in that certain problems are brought into focus and sharper, more insightful thinking should be stimulated to critically examine ways to improve predictive accuracy and to better understand the limitations for predictive accuracy of any habitat model.

FACTORS CONTROLLING FISH ABUNDANCE

One of the arguments used by defenders of IFIM is that WUA is directly related to the "carrying capacity" for a particular species and attempts to relate WUA to biomass is prone to error because a population is rarely at carrying capacity. A problem here is the definition of the elusive concept of carrying capacity and how it can be determined -- a problem without a universal solution. Putting the problem of the determination of carrying capacity aside, the associated problem is that the basic assumption of IFIM is that carrying capacity is completely controlled by physical habitat which, in turn, can be accurately represented in a model by measurements of depth, velocity, cover and substrate (the temperature factor is at least recognized and flow-temperature model is available to use with IFIM Physical Habitat Simulation System [PHABSIM]).

In high gradient streams of the Rocky Mountain region, characterized by great variation in annual flow, I would agree that most fish populations are more "habitat limited" rather than "food limited", although limitations resulting from interactions with other species may be a more powerful limitation than is the physical habitat or potential food supply. The assumption that carrying capacity of a fish population in a stream is always limited by physical habitat is certainly not universally true. A list of factors affecting fish population abundance in streams can be stated as follows.

Physical Factors

1. Flow regime
2. Habitat quality
3. Water quality (temperature, pH, turbidity, etc.)

Biological Factors

1. Food abundance and availability
2. Predation
3. Competition and all interspecific interactions
4. Movement, migration

Of these factors, IFIM can only attempt to relate "habitat quality" to "flow regime", and is limited to very few indicators of "habitat quality." Thus, again one can understand the limitations for a good correlation between WUA and species abundance or biomass.

In relation to abundance governed by food supply, I would cite the example of the Au Sable River, Michigan, the subject of a recent report I prepared for evidence in court (Behnke 1986). The Au Sable River is a famous trout stream but the fishery (mainly for brown trout) has declined from a standing crop (biomass) of about 150 pounds per acre to about 100 pounds per acre during the past 15 years. The stream is fed by ground water and maintains a relatively constant year-round flow (flow close to 100% of ADF year-round). No change in flow has occurred in historical times. The only change in habitat consisted of $250,000 of stream improvement structures placed in a nine mile section of the river in the 1970's to provide more "resting" habitat in an attempt to reverse the downward trend (it failed). The only change known to have occurred in the Au Sable River concerns water quality. A large state fish hatchery ceased operation and its nutrient-rich effluent into the river ceased, and the sewage effluent of the town of Grayling was diverted away from the river. This pollution abatement reduced nitrate-nitrogen levels by 70% in the Au Sable, which in turn decreased primary production and invertebrate production (food supply to the trout). There was no significant change in trout abundance except for fewer age IV and V fish, but the growth rate was significantly reduced so that the size of trout of any particular age was much less in comparison to the period of nutrient enrichment, and this resulted in the decrease in population biomass.

A more detailed and quantified example of increased trout production in a stream as a result of nutrient enrichment concerns Berry Creek, Oregon and the study of Warren et al. (1964) who enriched a test section of Berry Creek with sugar (sucrose). The sucrose produced a proliferation of bacterial slime which was fed upon by aquatic insects which greatly increased in abundance, increasing in turn, the food supply to the trout population. The intake of food by the trout population was doubled, but production of the trout population increased by more than seven-fold. This great increase in production is explained by an understanding of maintenance rations vs. growth rations. When a fish population is at or near the limits of its available food supply, most food is utilized for body maintenance (maintaining the status quo) and little is available for growth (production in the population is low). Once maintenance requirements are met, all additional food goes into growth. Thus, by only doubling the total food intake, production of the trout population increased more than seven-fold because the additional food produced from stream enrichment went into growth.

The lessons learned from the examples of the Au Sable River and from Berry Creek makes it clear that food cannot be ignored as a factor controlling population biomass. The physical habitat did not change in Berry Creek (WUA constant) and was improved in the Au Sable River by creating lower velocity areas with cover in the channel (WUA would have increased during time of biomass decline).

The studies of Hawkins et al. (1983) and Murphy et al. (1981) on small Oregon streams in the Cascade Mountains demonstrated a great increase in primary production (aquatic plant production), which led to increases in aquatic insect production, and increase in biomass of trout and salamanders after clear-cutting of the forest. This cause-and-effect relationship in this example is sunlight which drives primary production and which was essentially blocked from the streams by a complete canopy of vegetation before the clear-cuts. The dynamics of the energy flow in these streams, eventually producing food for the fish, changed from mainly allochthonous input (from surrounding terrestrial environment) to predominantly autochthonous (sunlight stimulating primary production in the stream) because of the removal of trees.

All of the above examples demonstrate that physical habitat, especially when interpreted only as depth and velocity, is only one of many controlling factors of fish biomass in a stream. They also reveal the range of variables and complexities that would have to be considered in an attempt to develop a stream habitat model to predict changes in biomass resulting from any environmental change.

There is undeniable evidence that physical habitat does exert a strong controlling factor on the abundance-biomass of a fish population. A section of Lawrence Creek, Wisconsin, was structurally changed to convert a predominantly wide, shallow, high velocity riffle area without cover in to a more narrow, deeper, low velocity area with overhanging bank cover (Hunt 1976). The increase in biomass of brook trout in this section after improvement approximately doubled, but the biomass of adult trout increased about four fold. This was due to the conversion of "juvenile" habitat into "adult" habitat. At the same time, it can be assumed that total invertebrate production decreased by the conversion of riffles into pools and decrease in absolute channel area after the improvement, but the improvement in habitat (deeper, slower water with cover) allowed more adult trout to more effectively utilize the food supply that was formerly underutilized. All similar types of stream improvement projects operate under the assumption that the fish population is habitat limited (at least within the section of the stream to be improved) and by creating deeper, low velocity areas with cover in areas lacking such habitat, the population will increase. It can not be assumed that the food supply will increase to any extent from fish habitat improvement, only that fish will now be able to better utilize the food that was not previously available to them. When habitat improvement has been effective for increasing fish biomass, then the above assumptions are proven correct. Where habitat improvement has not been effective, such as in the Au Sable River, then other factors, such as food are is limiting (or the habitat structures were poorly designed or placed in wrong sites).

Empirical evidence relating habitat to flow and ultimately fish biomass concerns spring creeks and regulated rivers. Spring creeks typically have stable year-round flows, are low gradient, nutrient-rich, relatively deep and with macrophyte vegetation. Besides insect life, most such streams have an abundance of crustaceans (typically gammarid amphipods). The habitats of spring creeks may be more comparable to a lacustrine (lake) environment than to a high gradient, rocky, highly fluctuating stream. Spring creeks (English chalk streams, Sand Creek, Wyoming -- The stream that provided the extreme biomass point for Binns' HQI model, and gave the model such good correlation between HQI and biomass -- and some noted spring creeks in Montana) have long been recognized as the ultimate in trout streams -- biomass of 500 to 700 pounds per acre or more and rapid growth rate of the trout. In spring creeks, virtually the whole channel, in relation to depth, velocity, and cover, would be rated at maximum values. Thus, the optimum habitat allows the trout population to expand to the limits of its food supply (to attain its "carrying capacity"). When dams regulate rivers by eliminating the peak flood flows and elevate the late summer base flow above natural levels, the resulting flow regime becomes somewhat similar to a spring creek and the trout population responds in a similar manner to the improved habitat conditions (lower velocity during run-off, greater depth during late summer). Some of the most famous trout fisheries in the West are the result of river regulation -- South Platte River, Frying Pan River, Gunnison River, Colorado; "Miracle Mile" of North Platte River and Bighorn River, Wyoming, and many other examples.

Other instructive examples of changes in fish populations correlated with habitat changes concerns habitat protection measures such as fencing livestock away from streambanks on overgrazed watersheds. Results have often been dramatic with several fold increases in trout populations after riparian vegetation is restored to the banks, the stream channel stabilizes, becomes more narrow and deeper with overhanging cover (Behnke 1979). Essentially, the natural changes in habitat improvement from livestock protection is similar to the artificial improvement of Lawrence Creek, Wisconsin, discussed above. A wide, shallow, high velocity stream without cover is converted into a more narrow channel with slower, deeper water with cover. There has not been a change in invertebrate production (actually invertebrates may decrease) or in flow, but only in physical habitat that results in a large increase in the fish population.

It would be useful if before and after studies of streams subjected to natural or artificial habitat improvement and habitat changes due to flow changes from river regulation were conducted to develop and test habitat models and the accuracy of their predictions on fish population change. A problem I foresee for any complex habitat model is that simple factors such as depth and velocity can be objectively recorded by anyone following a set of rules and using standard equipment, but factors such as "cover" is subjective and different workers may arrive at very different "cover" values for the same stream. Also when "cover" is the result of complex and interacting factors, its simple compartmentalizing into standardized units for modelling may result in large errors when applied in different areas. For example, Loar (1984) found brown and rainbow trout to be negatively correlated with "cover" as measured by IFIM. I do not believe that the trout deliberately avoided cover in the Appalachian streams investigated, only that the trout's concept of "cover" differed from the IFIM concept.

Why WUA influenced by "compartmentalized" cover ratings are not interchangeable between streams or even between different sections of the same stream can be understood by comparing a holistic interpretation of "cover" and a reductionist breakdown of "cover" into measurable units. The mind of an experienced angler makes a holistic interpretation of a stream in arriving at a decision to where to concentrate his efforts -- the sites to cast bait or lures that provide the most favorable opportunity to catch larger adult trout. The largest trout select the areas of the stream with the greatest "volume" and complex cover, such as a deep hole beneath a bank, upturned tree roots, below large boulders or log jams. Those large volume areas might be considered as "first class accommodations". Other areas of the stream channels with certain parameters of depth, velocity and cover might be given an equal WUA rating by quantitative measurements and following the rules of IFIM, but to adult trout they are "second class" living accommodations, which will be used by smaller, subdominant trout. In such situations, biomass per unit area of stream channel may greatly vary between first and second class habitats even though they have equal WUA.

A major factor controlling the abundance or presence or absence of a species that can not be assessed by any present habitat suitability index for IFIM (or HEP) is the presence of other species resulting in predation and/or interspecific competition. In a typical river drainage in the West, the smaller headwater streams can be expected to be inhabited by brook trout or sometimes native cutthroat trout.

The larger (5-6-7 order streams) stream channels in the drainage will typically have brown and/or rainbow trout. This distribution pattern of trout species within a drainage is repeated over and over throughout the West. Thus, no matter how much habitat for cutthroat trout or brook trout might be quantified in a large stream, these species cannot establish viable populations in competition with brown trout or rainbow trout in a large stream environment. There are some exceptions to the rule, and these exceptions provide an opportunity to gain new, useful knowledge. In the Rio Grande drainage of Colorado, the native Rio Grande sucker has been virtually completely replaced by the introduced white sucker. In the Salt and Gila river drainages of Arizona and New Mexico, the native _Gila_ chub, loach minnow and spike dace, do not occur (or do not maintain viable populations) in the presence of smallmouth bass. For any attempt to construct predictive habitat models for species such as the Rio Grande sucker, _Gila_ chub, loach minnow, and spike dace, the "exclusionary principle" regarding the presence of certain non-native fish species must be recognized to have overriding power over "habitat" for predicting abundance or presence or absence.

In some instances, certain habitat components may interact in complex ways to influence fish abundance and this will interfere with predictions based on neatly compartmentalized models. IFIM assumes that fish respond to the habitat components as independent variables. If the fish utilize depth with dependence on velocity, then the assumption is violated and errors introduced for computation of weighted useable area (WUA). The dependency between depth and velocity in relation to fish use of a stream section may be relatively common in "run" areas of a river channel where large boulders that can serve as protective cover are absent. In such areas, at low flow, depth may be adequate to optimum for a species but due to the lack of cover, the species may make little use of the area because of predator avoidance (particularly if fish eating birds and mammals are common). During periods of higher flow, higher velocities create turbulent surface flow, reflecting and refracting light to such an extent that fish cannot be seen from above the surface. At such times, the fish will utilize the area with suitable depths because of the turbulence created by higher velocities (a dependency between depth and velocity influencing "useable area"). This is just one example of problems faced when attempting to develop a simple predictive habitat model which attempts to abstract the key factors controlling a species well-being. With sufficient time, money and expertise, a relatively accurate predictive habitat model might be constructed for a narrow-niche species in a small, homogeneous site with few or no interacting species, but it is highly improbable that such a site-specific model would retain its predictive accuracy when tested in different environments with different interacting species.

IFIM AND THE FUTURE

It is now apparent that the early naive hopes of many that IFIM would revolutionize the field of impact assessment for changing flow regimes by its ability to accurately predict meaningful biological changes correlated with flow changes will not be fulfilled, mainly because the quantitative output, WUA, does not accurately correlate with biologically meaningful attributes of the target species such as numbers and biomass. I do not foresee IFIM fading away from the environmental assessment scene however, because, 1. it has a large advocacy group, 2. I know of no better methodology to replace it, and 3. it's usefulness can be greatly improved over past performance in relation to new additions and refinements and particularly in relation to the experience and expertise of the user.

The personnel of the U.S. Fish and Wildlife Service Instream Flow Group are aware of IFIM problems and are continually working on ways to improve predictive accuracy. In a recently published habitat suitability index model and instream flow suitability curves for brown trout (FWS/OBS-82/10.71), I note that the suitability curves used in the habitat model are classified into four categories: 1. (most commonly used) are based on data derived from literature and professional judgement ("canned" program); 2. Curves derived from site-specific original data (utilization curves); 3. Utilization curves corrected for environmental bias -- comparing "utilization" vs. availability to arrive at "preference" curves; 4. "Conditional preference curves" to take into account interaction among variables (as discussed above for depth and velocity). I also noted that the recent velocity curves differentiated between mean water column velocity and "nose" velocity (velocity at site where fish exists). This is an important distinction because in higher gradient streams the high average current velocity will result in low WUA for most of the stream channel, although boulders, logs, etc. creating turbulent flow with small areas of microhabitat with pockets of low velocity can allow for high utilization that would be overlooked in a straightforward recording of average velocities along a transect. This former lack of distinction between average velocity and nose velocity was likely the major reason for the poor performance of IFIM when tested in Montana trout streams. Nelson (1980 a.b.) concluded that: "The weighted useable area (WUA) values generated by the IFG incremental method for the rivers of the study do not provide an accurate index of the actual amount of habitat that is available for brown and rainbow trout at the selected flow of interest. As a result, the IFG flow recommendations for the five study reaches are unreliable." This example also demonstrates the importance of experience and expertise of the user. When it is recognized that it is not the average velocity that determines "useability" but the amount of microhabitat with pockets of low velocity amidst an area of high velocity, the IFIM procedure should be modified to quantify the amount of microhabitat.

The Colorado Division of Wildlife has had some success in its application of IFIM to fishery problems (Nehring and Anderson 1984, Anderson and Nehring 1985), and this success is due to the experience and expertise of Barry Nehring. Instead of simply obtaining data for instantaneous correlations with biomass, the CDOW studies concentrate on flows in relation to determination of year-class strength (flows during spawning, incubation, and for newly hatched fry) and survival into older age classes (overwinter flow). Limiting factors (minimal WUA values) are examined for their insight into the factors determining trout abundance, particulary in rivers below dams. Nehring and Anderson have now well-documented the range of flows determining high year-class strength and low year-class strength for most of Colorado's major regulated rivers. It could be argued that this could have been accomplished without the aid of IFIM, by simply using USGS flow records, adequate sampling, and common sense. This may be true, but as a vehicle for communication of complex fishery information to non-biologists (such as administrators in water agencies), a computer printout and WUA curves relating flow to good and poor year-classes are impressive and for getting a point across.

Mr. Nehring has also developed innovative ways to manipulate WUA data to provide additional insight into problems (Nehring and Anderson 1984). If IFIM were to have more users with the experience, expertise, enthusiasm, and insights of Mr. Nehring, greater credibility of this methodology would be expected in the future. I would not, however, expect that WUA will ever be a consistently accurate predictor of a species biomass in different environments.

RECOMMENDATIONS

The problem that this report is designed to overcome concerns unwise and unwarranted demands that may be made by uninformed persons during discussions and negotiations regarding potential environmental changes resulting from a change in flow regime. Unless the spokespersons representing various agencies are extremely knowledgeable about the river and its biological system, and also knowledgeable about assessment methods and methodologies, there is likely to be a reflex reaction requesting "baseline studies", IFIM or HEP analyses, etc., before the potential problems are clearly identified.

The right questions must be asked: What river (or section of a river) will be changed? What are the target species of concern? How will the future annual hydrograph differ from the past? How might this change affect the target species, negatively and positively? What opportunities are there for flexibility and enhancement measures?

Once the area, the target species, and the issues have been defined, the questions concerning the utility of an assessment methodology to predict impacts can be addressed. This will require people with a high level of knowledge to arrive at a best solution. For example, in the Verde-Salt River drainage it might be requested that IFIM analysis be made for spike dace and/or loach minnows. The important questions to ask in such a case would be: are smallmouth bass present in the river section of concern? Is there a single example where loach minnows or spike dace maintain viable populations in the presence of bass? How reliable might be any habitat model constructed for these rare species? If models were made and incorporated into PHABSIM to correlate habitat with flow changes, how predictive would they be? What would the WUA values correlate with in regards to something meaningful about the species? How useful would the WUA values be for decision making?

In relation to the "exclusionary principle" I would point out the problem illustrated in my February report on the lower Verde trout fishery. Two bald eagle nests on the lower Verde where eagles rear young each year, makes this endangered species the species of highest priority for any environmental assessment. The eagles eat carp, suckers and catfish -- flows optimizing eagle food exclude flows for trout from serious consideration.

The point to be made is, that for the well-being of the target species in the biological system, more than technician grade studies on laying transects and recording data is needed. Holistic interpretive synthesis by persons knowledgeable about the river, its past and proposed flow regimes and of the target species is necessary. At least the input of higher level expertise should identify critical areas to see that the transects are most correctly sited. Then, analysis of depth-velocity changes might provide insights into limiting factors and opportunities for enhancement. The hard questions concerning the precise purpose of any proposed analysis and the predictive accuracy expected from any analysis should be asked during the earliest stages of negotiations.

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Nehring, R. B. 1979. Evaluation of instream flow methods and determination of water quantity needs for streams in the state of Colorado. Report to U.S. Fish and Wildlife Serv., Coop. Instream Flow Group: 144p.

June 16, 1986

Dear Bob,

I agree with our statement on page 8 of Critique of Instream Flow Methodologies, which states, " I believe most who have been intimately involved with IFIM will agree with the theme of my critique, but perhaps not all of the details." I would say most of us (intimately involved with IFIM over the years) probably even agree with most of the details. I further agree that "--- in recent years--- too many people were captured by the 'illusion of technique'. They had a naive faith that confused objectivity and quantification with biological reality." The IFIM/WUA concept, as I understand it, has definitely been misused and abused by too many lazy biologists looking for a universal methodology to apply to their problems (work) and make it even less necessary to do any critical thinking or labor intensive work.

However, I believe the IFIM/WUA concept will (in the next 10-20 years) evolve into an incredibly complex and highly predictive modelling technique of almost incredible accuracy and precision, given a certain set of basic assumptions and criteria that precede the site-specific application.

The criticisms that you direct towards the IFIM/WUA concept are somewhat misdirected. I feel you are using the examples of places where the IFIM/WUA system was misused in the first place to support your contention that all modelling techniques are not very useful in predicting future biomass or numbers of fish, because of the incredible complexity of natural systems. The Au Sable and Berry Creek cases demonstrate my point. In these two cases the neglecting of the alterations in nutrient input into the lotic environment were certainly the overriding factors that were controlling the productivity of those streams. However, not including the nutrient factor in the IFIM/WUA evaluation does not make the IFIM methodology a useless approach, it only means that it was misused. Nutrients, sunlight, the presence or absence of mutually exclusive predators (such as the smallmouth bass in the Gila and Salt river systems) can easily be modelled into the IFIM/WUA methodology. Its just one more set of suitability of use curves, that in these cases, would override those of depth, water velocity, substrate, and temperature.

The power of computers properly coupled with such innovative concepts as IFIM/WUA will become increasingly more precise and accurate in the future. But the emphasis is on the words "proper use". Any predictive model can only operate accurately as long as the set of pre-conditions or limitations in the application is well defined, and not seriously violated.

Use the Gunnison River as a case in point. AS long as the Curecanti Reservoir system controls the flows, the temperature, and the nutrient input into the Gunnison River below Crystal Dam we are operating under a certain set of conditions that are, as you would say, (p.3) "---stable, isolated and highly recurrent---". Given these preconditions, the IFIM/WUA concept is extremely precise and accurate in correlating numbers of rainbow and brown trout produced by a given amount of fry WUA during the critical emergence period. I strongly believe that these correlations can be used to predict quite accurately the number of 1+ yearling rainbow and brown trout that are recruited to the population. The results are quite precise, accurate, and quantifiable. As such, the IFIM/WUA concept is an extremely useful tool in the mitigation arena. Furthermore, for rainbow and brown trout at least, the concept can be transferred to other environments, i.e., for example on the South Platte River, to determine mitigation flows (by month) below Two Forks, should that project ever become a reality.

I like the way you close the paper on p. 17-18, and I heartily agree with you. "I do not see IFIM fading away from the environmental scene---", for all the reasons you state. But I would disagree with you; I do expect that WUA will be a consistently accurate predictor of species biomass and numbers in different environments (within a set of bounds and environmental constraints that are pre-conditions that define the application of the IFIM/WUA model).

Certainly, IFIM/WUA is not a panacea to be universally used in all situations. But it is, and will become more so, and increasingly useful tool to quantify the potential impacts of many types of environmental alterations and perturbations in stream ecology.

All in all, the report is very well balanced from the total perspective, pretty much right on target. You are a highly respected guru in the aquatic resource field. You use your "clout" in a very judicious altruistic manner. It is refreshing to see and know someone who still calls "a spade a spade" irregardless of who is paying the bill. Keep up the good work.

On a personal note I really feel you give me far too much credit, too much "positive press", too much of the time. Rick Anderson was a co-equal on this project for five years and deserves much more credit than he gets, and I get far more than I deserve. Rick is just a quiet, unassuming, but very intelligent and extremely competent biologist. When you cite us (as you did in your critique) please give Rick the consideration due him as well as a part of the team that we were.

Finally- I reviewed the Au Sable report as well. Excellent analysis, excellent writing, extremely well done. It should have made Alexander and Clark feel good to have you come down on their side of the situation.

Keep in touch. I always look forward to trips to "the FORT" when I get a chance to touch base with you and Dick Klein and continue the dialogue of "friendly persuasion".

Sincerely,

Barry

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SRP **Memo**

Date May 1, 1987

TO Mark DeHaven, Bob Behnke, Mikel Moore, Fritz Beeson

FROM Bill Warskow

RE: PENDING ARIZONA INSTREAM FLOW REGULATIONS

Paul J. Barrett, Biologist, U. S. Fish and Wildlife Service, sent me a copy of his and Marty Jakle's draft of "Survey of Instream Flow Methods for Use in Arizona." Mark and Bob, please add this to your list of materials forwarded to you on April 15, 1987 for your review and comment. Because you probably will not receive this by the May 4, 1987 deadline identified in my April 15 memo, I will not expect your comments on this document until May 11, 1987.

Mike and Fritz, this document is being sent to you for information only and no comments from you are required.

Bill

Bill Warskow

BW:njs
Enclosure

DRAFT

[Ca 1987]

Survey of Instream Flow Methods for Use in Arizona

by

Paul Barrett [1/] and Martin D. Jakle [2/]

In December 1986, the Arizona Department of Water Resources (DWR) convened a task force of technical experts to advise them in the area of instream flow. DWR has concluded that instream flow is a legal use of water if associated with recreational or fish and wildlife uses (Anonymous, 1983). A specific goal of the task force is to recommend to DWR acceptable methods of quantifying instream flows. This paper is a result of that request. We wanted to determine what methods are available and then select those which best suit existing needs. We have reviewed the different methods which have been, or are currently being used to quantify instream flows, given a brief description of each, and commented on their applicability to Arizona's streams.

It should be noted that most of the research and implementation of these methods have been by Federal and State research groups and management agencies. As a result, many of the results and descriptions have remained unpublished or have been published in government information publications which may be difficult for many investigators to locate. Hopefully, this paper will expose these techniques to a wider audience.

Basically, the methods for quantifying instream needs fall into two categories, hydrographic and hydraulic. Hydrographic methods are based on a hydrograph of the streams flow. Set percentages of this flow, for example, 60 percent of the mean annual flow are reserved for instream flow uses. Perhaps the best known method in this group is the Tennant Method and its subsequent modifications.

Hydraulic methods use data which are gathered at specific stream locations. Measurements such as stream depth, velocity, width, and water surface elevation are taken. A mathematical formula is then applied to these data and the stream is modeled at different flows. Biological data such as water velocity necessary for spawning is married with the hydraulic data and changes in habitat (primarily fish habitat) are determined for different flows. The best known methodology in this group is the Instream Flow Incremental Methodology (IFIM) and it too has several variations.

Methods will be presented primarily in chronological order.

One Flow Method

This method was used as early as 1963 in Oregon (Sams and Pearson, 1963; Stahlnaker and Arnette, 1976; Weshe and Rechard, 1980). It recommends an optimum flow based on calculations from aerial photographs of average stream width and pool width. As Weshe and Rechard (1980) point out, it can only be

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used on streams which are large enough to accurately evaluate widths from aerial photographs. Measurements of mean depth and velocity are taken over redds of the species concerned. The mean pool width is multiplied by the mean depth and mean velocity taken over redds to yield the recommended flow.

This method assumes that spawning flows are the limiting for the species of concern. Additionally, the method should only be used when the mean stream width is close to the mean pool width. With the advent of techniques having wider applications and more rigorous ones, including the recommendation of "minimum" as opposed to optimum flows, this method has apparently fallen out of use.

Hoppe and Finnell Method

From 1969 to 1970, the life history requirements of several fish species were collected on the Fryingpan River in Colorado and compared with average annual flow records for this period (Hoppe and Finnell, 1970; Bayha, 1978; Weshe and Rechard, 1980). Average daily flows for the period of record were ranked from highest to lowest, and a flow duration curve (discharge versus percentage of time flow is equaled or exceeded), was constructed. By comparing the biological data to the flow data it was determined the flushing flows over a 48 hour period were adequate at the 17th percentile level and above, i.e., the flow which is equaled or exceeded 17 percent of the time. Spawning flows were adequate at the 40th percentile and food production and cover at the 80th percentile. It should be stressed that their data were developed only for the Fryingpan River, Colorado, and as such should be consider site specific. This method could not be applied to Arizona's streams without first collecting biological data.

Flow Frequency Analysis

In this technique, stream flow records are examined to determine the lowest average flow for seven consecutive days for 10 year periods. This low flow is then recommended as the minimum instream flow. This technique was originally intended for use in overdesigning treatment plants so water quality standards could be maintained at low flows. Subsequently, it was adapted to fisheries flows, but has never been widely accepted (Bovee, pers. comm.). Obviously, flows based on this technique are very severe and have little basis in biological fact.

Tennant Method

One of the first techniques developed and one of the most widely used is the Tennant or Montana Method (Tennant, 1976). Between 1964 and 1974, Donald Tennant examined 11 streams in Nebraska, Wyoming and Montana. These streams represented a variety of stream morphologies and he documented physical, biological and chemical changes in the streams over the range of their annual flow fluctuations. Tennant concluded that changes in aquatic habitats are extremely similar among streams having similar annual flow regimes. He then surmised that 10 percent of the mean annual flow (MAF) would sustain short-term survival for most fish species. To sustain good survival habitat, 30 percent of the MAF was needed. Sixty percent of MAF provided excellent habitat. Tennant then proposed a range of percentages of the MAF regime to maintain desired flow conditions on a seasonal basis (Table 1).

Health of Habitat	Percent of Mean Annual Flow (MAF) October to March	April to September
Flushing or Maximum	200% MAF	
Optimum	60% to 100% MAF	
Outstanding	40%	60%
Excellent	30%	50%
Good	20%	40%
Fair	10%	30%
Poor	10%	10%
Severe Degradation	less than 10%	

Table 1. Instream flows necessary to maintain habitats, from Tennant (1976).

To use the Tennant method, the MAF for a stream is determined. These data can usually be calculated from U.S. Geological Survey (USGS) data using the period of record for the stream. Tennant recommends that the stream be visited to observe, photograph, sample and study the flow regimes at approximately 10, 30, and 60 percent of the MAF. The investigator can then adjust the recommended flows where necessary.

The Tennant Method has the advantage of being fast and easy to accomplish if stream flow data exist for the stream in question. However, this method °relies heavily on the professional judgement of the investigator. As with most methods, stream flow recommendations are instantaneous flows, which means that stream flows should meet or exceed the recommended flow at all times.

Modified Tennant Method

Several modifications to Tennant's method have been proposed. The application of the Tennant Method in its strictest form will result in unprecedented low flows (only subsurface flow in extreme cases) in streams with extremely constant base flow conditions throughout the year. In 1976 Tennant suggested adding the words "or natural streamflows if less than recommended minimum" after each recommendation. Additionally, he recommended 30 percent of the MAF as the "minimum flow" to protect aquatic resources, although flows as low as 10 percent of the MAF could be used as the absolute minimum, e.g., short term survival flows. He also suggested reconstructed virgin flows should be used to determine MAF.

Bayha (1978) also suggested that spring flushing flows may be required to clean spawning gravels, recharge wetlands and aid in fish spawning migrations in some areas. In locations where this may be an issue, it was recommended that 100 percent of the average annual discharge during the normal spring runoff be maintained.

Hilgert (1982) applied Tennant's percentages to streams in the Sandhills region of Nebraska. Instead of using the average annual flow, he used an estimate of base flow for making instream flow recommendations. Base flow was estimated by using the median flows during the dry season, e.g., November, December, and January. This method would remove a fixed percentage of water thoughout the year and high flow months would be treated the same as low flow months.

South Dakota State Method

The question of siltation was raised by (Tessman, 1980) State University's Water Resources Research Institute South Dakota. They attempted to parallel the natural flow regime during the yearly cycle, but also recognized that low flow periods as well as high flow incidents are important in maintaining a biotic community. Assuming that streams are more sensitive to pertubation during periods of low flow, they used the mean monthly flow as a level beneath which no water should be removed during months of low flow. The two six month periods, April through September and October through March, are used in the Tennant method; however, these periods do not correspond well with the high and low flow periods respectively in prairie streams. They, therefore, recommended applying a compromise value of 40 percent of the mean annual flow with the following stipulations:

> "Extreme fluctuations in periodicity are accommodated by applying a compromise value of 40 percent on a monthly basis, with some stipulations. During low water months, when the mean monthly flow is less than 40 percent of mean annual, the mean monthly flow is designated as the minimum flow. This preserves flow of low water months. Since a mean flow value is used, there will obviously be months when the actual runoff is less than mean runoff. The mean monthly flow simply serves as a constraint to indicated that no water may be abstracted if actual flow is equal to, or less than mean flow. It is not a specification that minimum flow must be maintained at the mean monthly flow because flows of this magnitude cannot be expected in most years. If the mean monthly flow exceeds 40 percent of the mean annual, but 40 percent of the mean monthly is less than 40 percent of the mean annual, then 40 percent of the mean annual is designated as the minimum monthly flow. If 40 percent of mean monthly exceeds 40 percent of mean annual, then minimum monthly is 40 percent of mean monthly. A summary of this procedure follows:

Situation	Minimum Monthly Flow
1. mean MF <40% mean AF	mean MF
2. mean MF >40% mean AF and 40% mean MF <40% mean AF	40% mean AF
3. 40% mean MF >40% mean AF	40% mean MF

MF = Monthly Flow
AF = Annual Flow

> "Further, a 14-day period of 200 percent of mean annual flow is specified during the month of highest runoff for purposes of flushing the stream's silt load and flooding stream side habitat. By using this modified procedure,

the annual periodicity may be mimicked without gross over appropriation of flow that would result by strict application of the Montana Method."

This description reads somewhat like a tax form; however, the results of this method are shown in tables 2 and 3, and shown graphically in Figures 1 and 2.

Great Plains Method

The Great Plains Method or Modified Montana Method is another variation of the Tennant Method and was proposed in 1974 by a group working in the Northern Great Plains Resource Program (Bayha, 1978). It was based on the premise that water presently flowing in a river represents flows supporting present levels of aquatic or related resources, i.e., the biota of the river is in equilibrium. They suggested that for each month, the flows be ranked from highest to lowest. The upper and lower 15 percent were then eliminated. A flow duration curve was then constructed using the remaining data. An instream flow recommendation of a 90 percent exceedence flow (i.e., 90 percent of the time the flow will be greater than the stated value) is recommended for each month. These recommendations can then be adjusted for tributary inflow to and diversion out flow from the waterway in question if the stream gage is not ideally located. Finally, these values can also be adjusted for the specific fishery resources and specific species life stages in the area, e.g., blue ribbon trout spawning areas.

This method has the advantage of being fairly fast and easy and takes into account the stream's annual cycle in that it is based on monthly data. It may be somewhat subjective in that it makes general assumptions about the amount of stream flow necessary for a healthy aquatic community.

New England or Connecticut Method

This method is based on the premise that flows in June are optimum for fisheries and desirable flows for the rest of the year can be related to the June flow. Additionally, nearby watersheds should exhibit similar flow patterns and results are applicable to streams with approximately the same flow magnitudes as the stream from which the data were gathered. In some cases the results were also applied to streams having significantly different flow magnitudes (Bayha, 1978).

The average monthly median and lowest flows are determined using USGS stream gaging records. After establishing the average median June flow, the flows for the other 11 months are calculated as a percentage of June flows as follows: July through September, 90 percent; October through February, 110 percent; March through May, 180 percent. Minimum flow is established by setting the minimum June flow to 30 percent of the June average median flow and apportioning the others from this flow as above.

If the drainage area in square miles at a selected gage is known, the flow data can be converted to cubic feet per second per square mile of drainage area (cfs/mi^2). The recommended optimum and minimum fisheries flows for nearby similar streams can then be calculated by multiplying the 100 percent and 30 percent June cfs/mi^2 by the drainage area of the nearby basin.

Figure 1. Comparison of different hydrographic instream flow methods and the median flow based on stream flow data at USGS gage number 5085, Verde River at Tangle Creek.

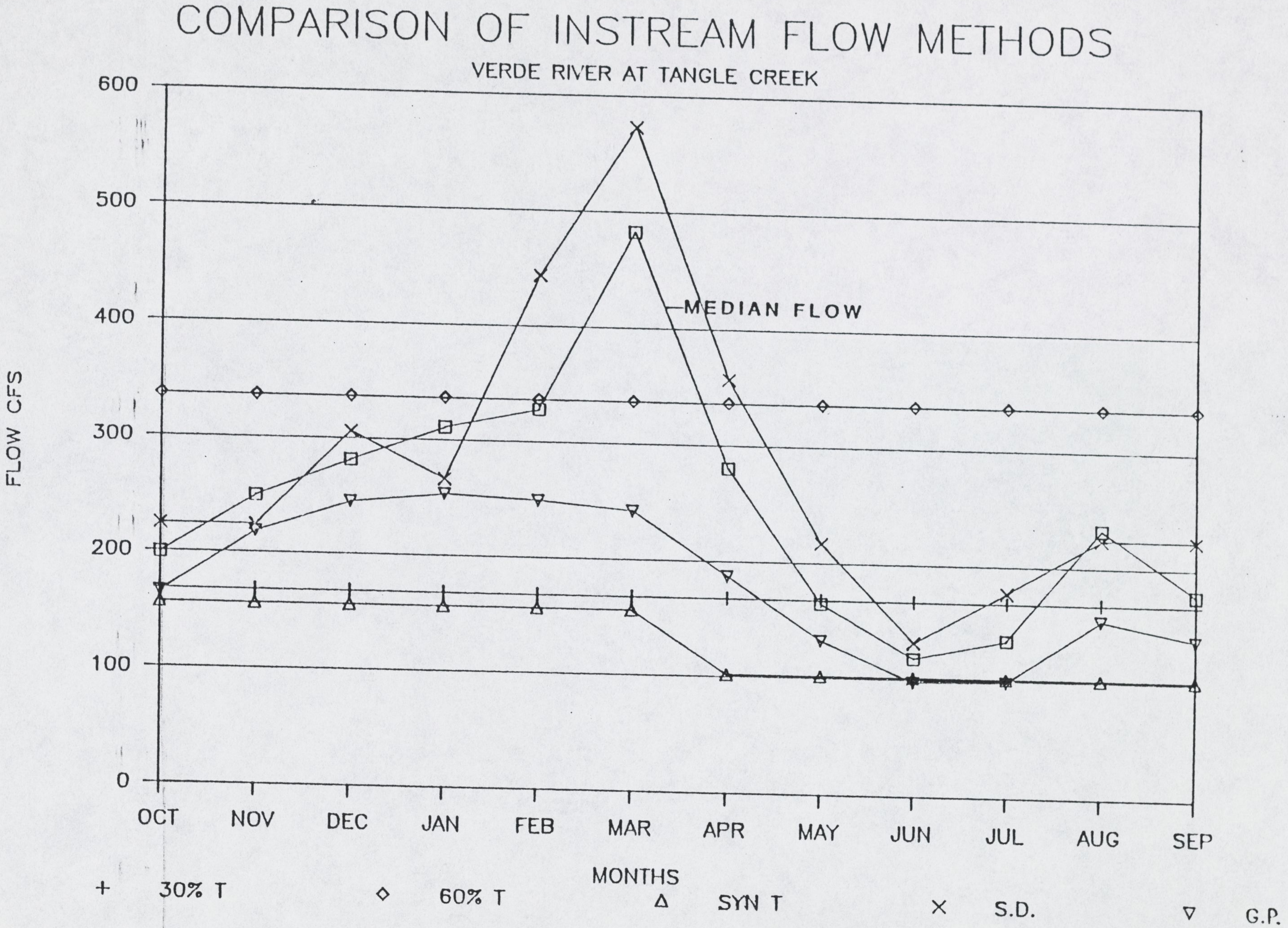


Figure 2. Comparison of different hydrographic instream flow methods and the median flow based on stream flow data at USGS gage number 5037, Verde River at Paulden.

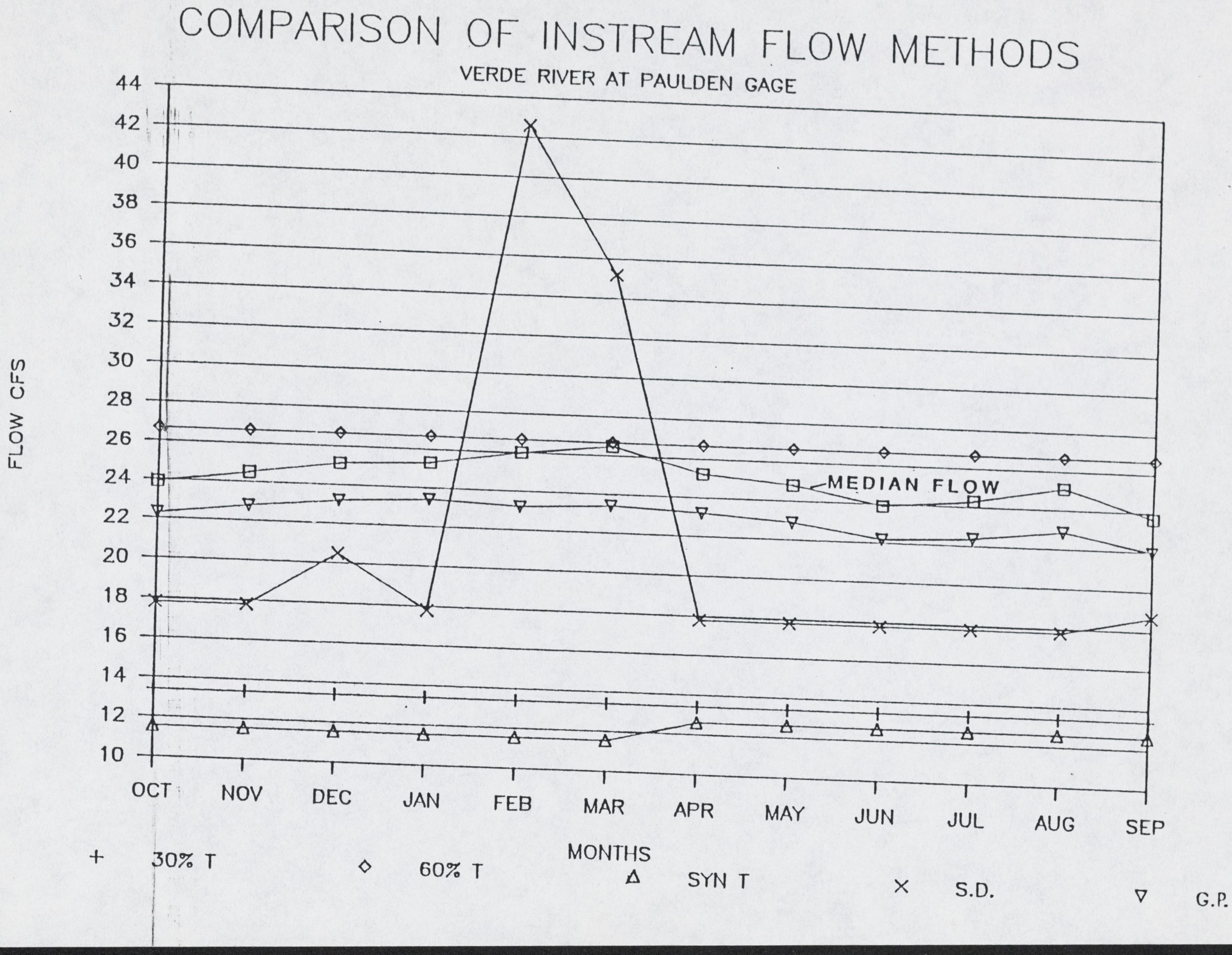


TABLE 1. COMPARISON OF DIFFERENT INSTREAM FLOW METHODS
BASED ON STREAM FLOW AT USGS GAGING STATION 5085--VERDE RIVER AT
TANGLE CREEK. DATA TAKEN FROM USBR REPORT BY TOM MYERS, 1986

MONTH	MEAN FLOW CFS	MEDIAN FLOW CFS	HISTORIC MINIMUM	TENNANT 30 %	TENNANT 60 %	SYNTHETIC TENNANT	MODIFIED TENNANT	S.DAKOTA METHOD	GRT PLAINS METHOD
OCT	351.0	199.0	61.0	168.3	336.6	156.4	42.0	224.00	165.00
NOV	367.0	249.0	61.0	168.3	336.6	156.4	42.0	224.00	218.00
DEC	766.0	281.0	61.0	168.3	336.6	156.4	42.0	306.00	245.00
JAN	665.0	311.0	61.0	168.3	336.6	156.4	42.0	266.00	253.00
FEB	1110.0	328.0	61.0	168.3	336.6	156.4	42.0	444.00	249.00
MAR	1434.0	483.0	61.0	168.3	336.6	156.4	42.0	573.00	242.00
APR	892.0	280.0	61.0	168.3	336.6	135.6	42.0	357.00	187.00
MAY	217.0	165.0	61.0	168.3	336.6	135.6	42.0	217.00	133.00
JUN	133.0	119.0	61.0	168.3	336.6	135.6	42.0	133.00	98.00
JUL	177.0	136.0	61.0	168.3	336.6	135.6	42.0	177.00	99.00
AUG	339.0	233.0	61.0	168.3	336.6	135.6	42.0	224.00	154.00
SEP	276.0	177.0	61.0	168.3	336.6	135.6	42.0	224.00	138.00
ANNUAL	561.00	246.75							

BASE FLOW = THE MEAN OF THE MEDIAN FLOWS FOR MAY, JUN & JUL

TABLE 2. COMPARISON OF DIFFERENT INSTREAM FLOW METHODS
BASED ON STREAM FLOW AT USGS GAGING STATION 5037--VERDE RIVER AT
PAULDEN. DATA TAKEN FROM USBR REPORT BY TOM MYERS, 1986

MONTH	MEAN FLOW CFS	MEDIAN FLOW CFS	HISTORIC MINIMUM	TENNANT 30 %	TENNANT 60 %	SYNTHETIC TENNANT	MODIFIED TENNANT	S. DAKOTA METHOD	GRT PLAINS METHOD
OCT	33.7	23.9	16.0	13.4	26.7	11.5	7.5	17.80	22.30
NOV	26.0	24.5	16.0	13.4	26.7	11.5	7.5	17.80	22.80
DEC	51.2	25.1	16.0	13.4	26.7	11.5	7.5	20.50	23.20
JAN	40.5	25.3	16.0	13.4	26.7	11.5	7.5	17.80	23.40
FEB	106.5	26.0	16.0	13.4	26.7	11.5	7.5	42.60	23.20
MAR	88.0	26.5	16.0	13.4	26.7	11.5	7.5	35.20	23.40
APR	35.8	25.2	16.0	13.4	26.7	12.6	7.5	17.80	23.20
MAY	24.4	24.8	16.0	13.4	26.7	12.6	7.5	17.80	22.90
JUN	23.6	23.9	16.0	13.4	26.7	12.6	7.5	17.80	22.20
JUL	26.0	24.3	16.0	13.4	26.7	12.6	7.5	17.80	22.40
AUG	33.6	25.1	16.0	13.4	26.7	12.6	7.5	17.80	22.90
SEP	45.4	23.7	16.0	13.4	26.7	12.6	7.5	18.70	22.00
ANNUAL	44.56	24.86							

BASE FLOW = MEAN OF MEDIAN FLOWS

Again, this method would be fast and easy to use on gaged streams, however, it assumes average median June flows are optimum which may or may not be true in Arizona.

Wetted Perimeter

The most basic of the hydraulic methods is the Wetted Perimeter Method. There are several variations of the method, but all must assume that maintenance of suitable conditions over riffles will maintain suitable conditions in other areas as well and that the wetted perimeter (figure 3) is closely related to fish habitat.

In all variations a critical cross section over a riffle is established. Next, the wetted perimeter of the cross section is calculated at various flows either through actual physical measurements at these flows, or through hydraulic simulation based on one set of physical measurements then using a computer model such as IFG 1 (Milhous 1978; Annear and Cander 1984). Wetted perimeter versus discharge is then plotted. The plot is examined and an inflection point in the curve where small decreases in discharge yield large decreases in wetted perimeter is the recommended instream flow (figure 4).

In addition to the above-mentioned assumptions, this method is based on the subjective judgement of the investigator reviewing the wetted perimeter versus discharge curves. Also, several inflection points may appear on a single curve complicating the flow recommendation decision. Conversely, there may be no distinct inflection point and the decision becomes more subjective.

Usable Width Method

The Usable Method builds on the Wetted Perimeter Method. This method was developed for salmonids, but different activities can be evaluated for other species of interest. It was one of the first methods which incorporated biological parameters directly into the analysis (Thompson, 1972). Also known as the Oregon Method, it was originally developed for passage analysis in the Northwest (Bovee pers. comm.), but eventually was expanded to include spawning, rearing and incubation as well.

Criteria were developed for each species, usually using depth and velocity as variables, e.g., for chinook salmon, minimum depth (D_{min}) = 0.8 ft, and maximum velocity (Vmax) = 8 ft/sec. Next a single cross section was measured at multiple discharges or modeled at one discharge similar to the wetted perimeter method. Data were gathered and criteria established for the different biological activities the investigators thought were important such as passage, spawning, incubation, migration and rearing. Shallow bars were considered the limiting factor for passage. Single transects were placed across the shallowest portion of one or several bars. The percent of the width of the stream meeting a previously determined depth criteria was then calculated. A minimum flow recommendation can be made which will allow passage of adult fish over shallow areas. A similar method is used for spawning with transects being placed across spawning bars. The recommended discharge is that which creates suitable flow conditions over 80 percent of the bar during spawning season.

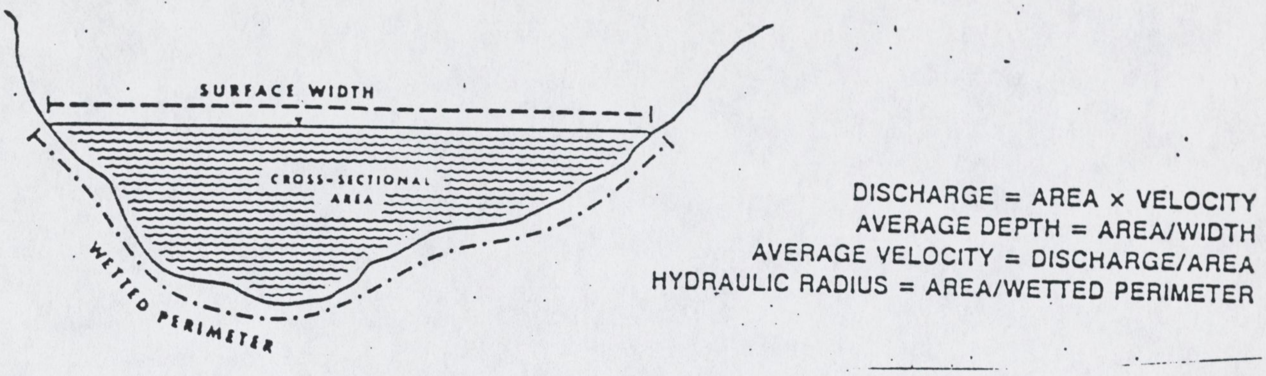


Figure 3. Average stream parameters calculated by Single Cross-Section Program (from Hilgert, 1982).

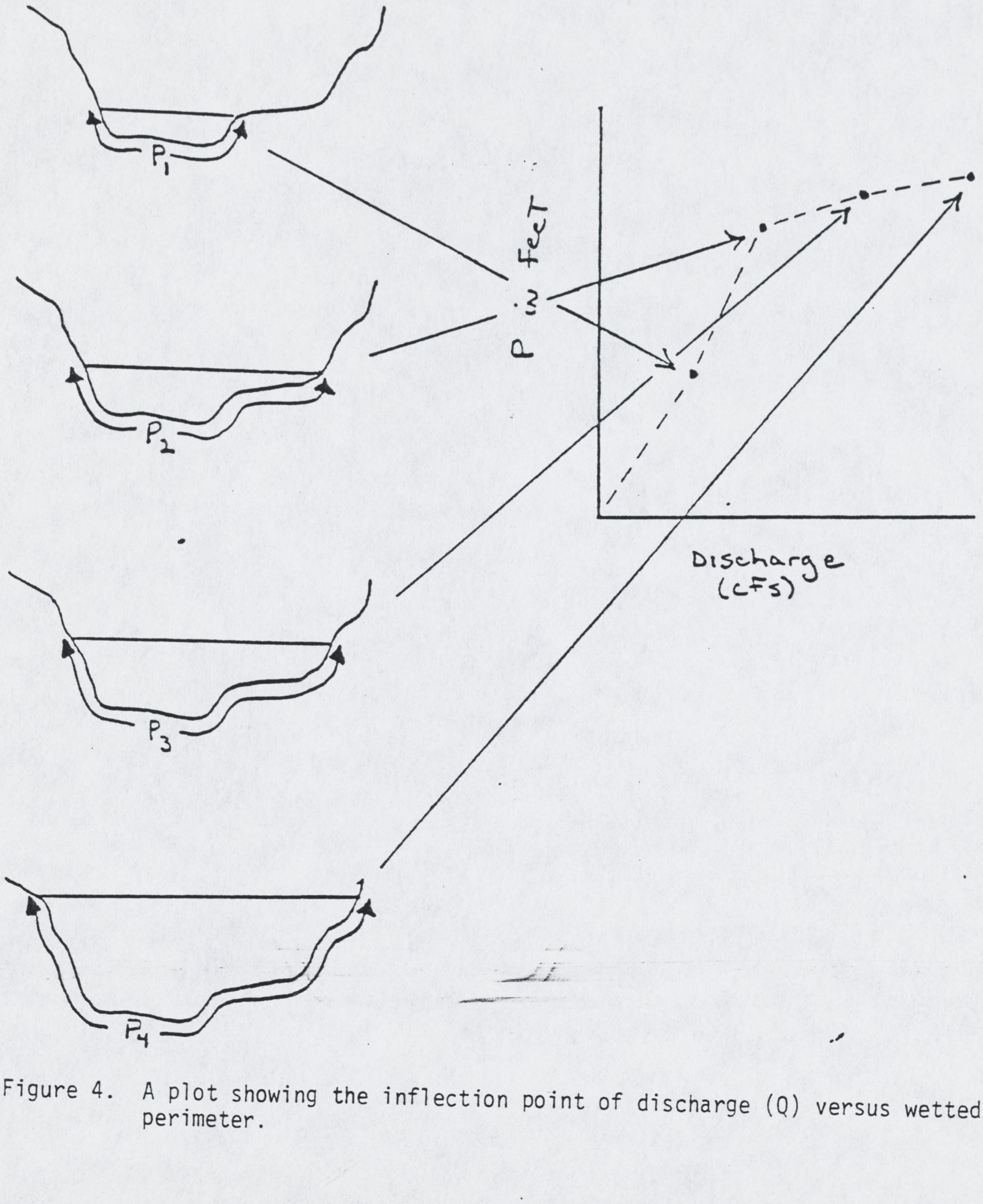


Figure 4. A plot showing the inflection point of discharge (Q) versus wetted perimeter.

Stream transects were not used to develop flow recommendations for rearing fish. Instead, the stream was studied during several flows and it was estimated which flows would be suitable based on the following six criteria:

1. Adequate depth over riffles.
2. Riffle/pool ratio near 50:50.
3. Approximately 60 percent of the riffle area covered by flow.
4. Riffle velocities 1.0 to 1.5 feet per second (fps).
5. Pool velocities 0.3 to 0.8 fps.
6. Most stream cover available for shelter for fish.

The incubation requirements were initially an estimate of the flow which would cover gravel areas used for spawning and create an intra-gravel environment conductive to successful egg incubation and fry emergence. Weshe and Rechard (1980) state this generally amounts to about 2/3 of the spawning flow. In 1974, Thompson related the amount of dissolved oxygen available at different flows to the success of salmon eggs making flow quantifications for this life stage less subjective.

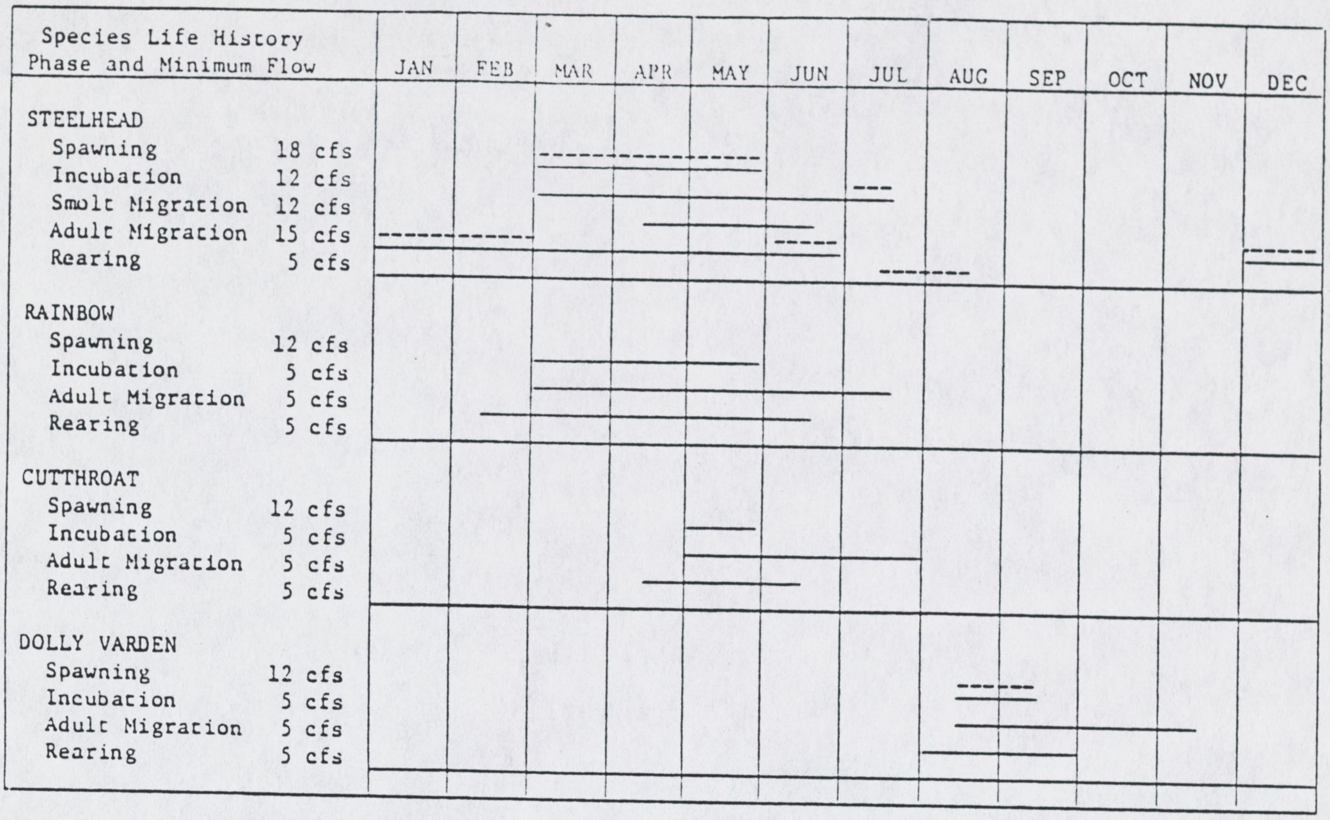
After flow recommendations are determined for each biological activity for the species of interest, a chart of the life history of each species and the recommended flow for each life stage is compiled (figure 5). The flow recommended for any two week period is the highest flow required to accommodate any of the biological activities which occur during that period.

This method has been modified by weighing the value of the stream velocities for each activity for each species (Sams and Pearson, 1963; Wesche and Rechard, 1980). The technique is identical except that a table weighing the useable velocities of each activity of each species is constructed (Table 4) and used to weigh the velocities determined across a transect by field measurements. The width which contains a particular velocity is multiplied by the corresponding weighing factor from the table and the sum of all these weighted values across a transect are used to determine the weighted usable area of the habitat which the transect represents at the flow.

<u>Idaho Method</u>

The Idaho Method combined the Usable Width Method with the Bureau of Reclamation's (Reclamation) Water Surface Profile (WSP) computer program and was developed for use on large rivers by White and Cochnauer (1975). The WSP program was originally formulated by the Reclamation to predict stream stages at flood flows. WSP calculates water surface elevations (WSLs) upstream from a control such as a gravel bar or constriction in the channel, based on energy losses between two adjacent stream cross sections. It is based on the concepts of mass balance and energy balance and relies upon an estimate of the friction or roughness of the stream bed, Manning's "n". The WSP computer program will model detailed depth and velocity predictions for a variety of stream flows based on one set of field measurements.

As with all models, WSP has limitations. Calibration of this model is difficult in streams with slopes exceeding two to three percent. Also, there must be a constant flow when collecting a set of field measurements because the model does not account for gains or losses of flow between transects. All hydraulic controls must be included to accurately run the model and finally,



Recommended Minimum Flow Regimen	JAN	FEB	MAR	APR	MAY	JUN	JUL	AUG	SEP	OCT	NOV	DEC
	15	15	18	18	18	15	12/5	5/12	12/5	5	5	15

Figure 5. Life history periodicity and minimum flow regimen for existing salmonid populations in Reynolds Creek, John Day Basin (from Thompson, 1972).

Species	Velocity Category (cm/sec)	Velocity Category (ft/sec)	Weight Factor
Spring Chinook Salmon (O. tschawytcha)	7.6-19.8	0.25-0.65	0.1
	19.8-22.9	0.65-0.75	0.4
	22.9-25.9	0.75-0.85	0.8
	25.9-56.4	0.85-1.85	1.0
	56.4-59.4	1.85-1.95	0.9
	59.4-62.5	1.95-2.05	0.6
	62.5-68.6	2.05-2.25	0.5
	68.6-77.7	2.25-2.55	0.2
	77.7-86.9	2.55-2.85	0.1
Fall Chinook Salmon (O. tschawytcha)	27.4-33.5	0.90-1.10	0.2
	33.5-42.7	1.10-1.40	0.6
	42.7-67.1	1.40-2.20	1.0
	67.1-76.2	2.20-2.50	0.4
	76.2-88.4	2.50-2.90	0.2
	88.4-94.5	2.90-3.10	0.1
Coho Salmon (O. kisutch)	13.7-16.8	0.45-0.55	0.1
	16.8-25.9	0.55-0.85	0.2
	25.9-32.0	0.85-1.05	0.6
	32.0-38.1	1.05-1.25	0.8
	38.1-50.3	1.25-1.65	1.0
	50.3-59.4	1.65-1.95	0.8
	59.4-68.6	1.95-2.25	0.4
	68.6-74.7	2.25-2.45	0.2
	74.7-93.0	2.45-3.05	0.1
Steelhead Trout (S. gairdneri)	36.6-45.7	1.20-1.50	0.4
	45.7-73.2	1.50-2.40	1.0
	73.2-85.3	2.40-2.80	0.6
	85.3-103.6	2.80-3.40	0.2

Table 4. Spawning velocity categories and weighting factors for weighted useable width analysis(from Sams and Pearson, 1963).

although only one set of measurements is needed, more are desirable. For a further explanation of both the theory and limitations of the WSP program, refer to Bovee and Milhous, 1978.

As in the original Usable Width Method, transects are placed across critical habitats for the various biological activities or life stages of interest. However, the WSP program is then used to predict WSL and the subsequent depth and velocity measurements at a variety of flows. These predictions are then used in place of a series of measurements at different various flows to develop flow recommendations.

Region 6 of the Forest Service (FS) used another variation on the Usable Width Method to determine optimum flow which is defined as the flow with the greatest amount of usable habitat (Swank, 1975; Swank and Phillips, 1976). Permanent transects were placed across typical channel cross sections representing spawning, rearing and food production habitats. Depth and velocity measurements were made across the transects for at least three flows, ranging from the lowest expected to the highest expected. Usable width is then calculated based on previously established species use criteria developed from literature searches or field data. On a single graph, a plot of usable width versus flow is drawn for each life function (Figure 6). Optimum flow will depend on the magnitude of the usable width differences between the curves and may be expressed on a range. A flow duration curve can be drawn to estimate the percentage of time flows within the optimum range can be expected. This method gives optimum flows but is quite labor intensive considering the accessibility of computers.

Habitat Mapping

This procedure was first developed to evaluate spawning habitat in Washington circa 1972 (Collings, 1972), and is also known as the Washington Method. First, multiple transects are used to map spawning area. Measurements of depth (D), velocity (V), and substrate (S) are made across the transect at each flow. Contours of equal D, V, and S are drawn on separate maps (Figure 7a) and then areas of suitable D, V, and S are delineated on these maps yeilding a composite map of all three variable maps (Figure 7b). The areas meeting all necessary criteria are measured using a planimeter. This process is repeated for a range of flows and a resulting plot of spawnable area versus flow is constructed (Figure 8). A minimum spawning flow was set on the flow providing 75 percent of the maximum possible spawning habitat. Like the previous method, this method is very time consuming considering recent computer software programs have been developed which accomplishes essentially the same thing.

Indicator-Species Overriding Consideration Method

The Indicator-Species Overriding Consideration Method (Bovee, 1975) is a technique which combines both the Usable Width and Habitat Mapping Method. The species with the narrowest range of discharge tolerances are used as indicator species for the entire system. It is assumed that if their needs are met, the needs of all other species in the system will likewise be met. Three biological functions were used; migration, spawning and rearing. This method was developed in the northern Great Plains and the species used as indicators were the paddlefish (*Polyodon spathula*) and sauger (*Stizostedion canadense*).

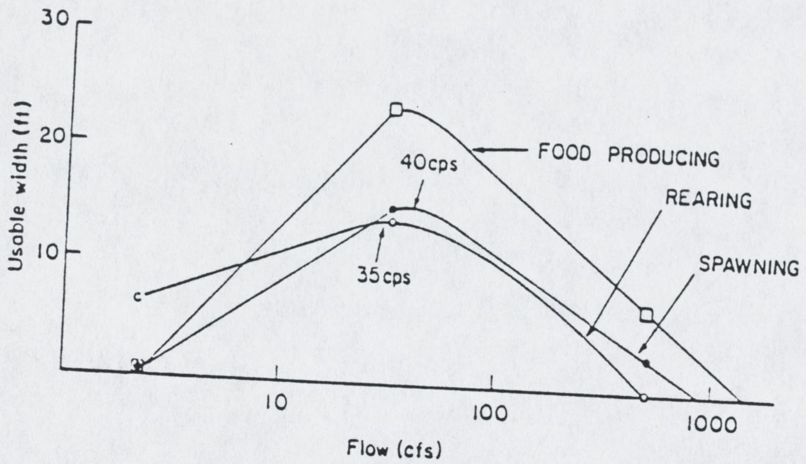


Figure 6. North Fork of the Crooked River instream flow analysis(from Swank 1975).

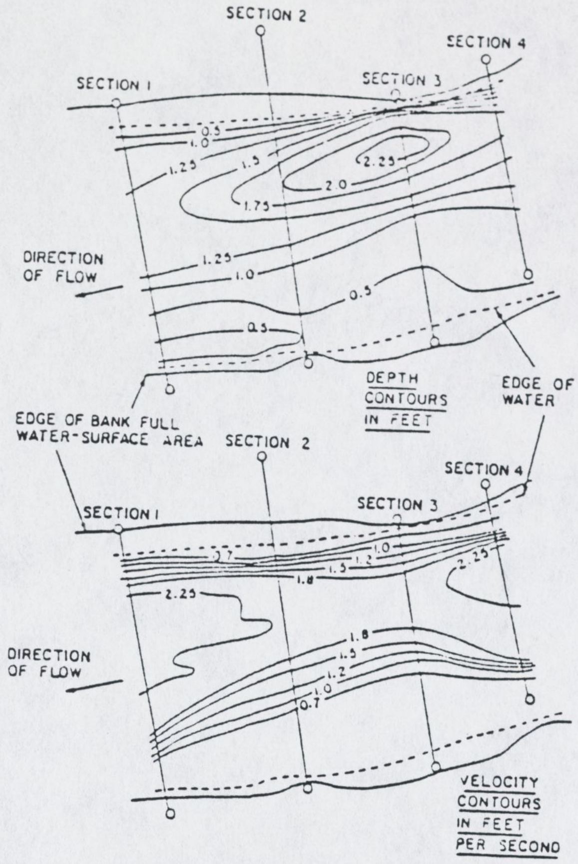


Figure 7a. Example of study reach water depth and water velocity contouring for one river discharge, North Nemah River(from Collings, 1972).

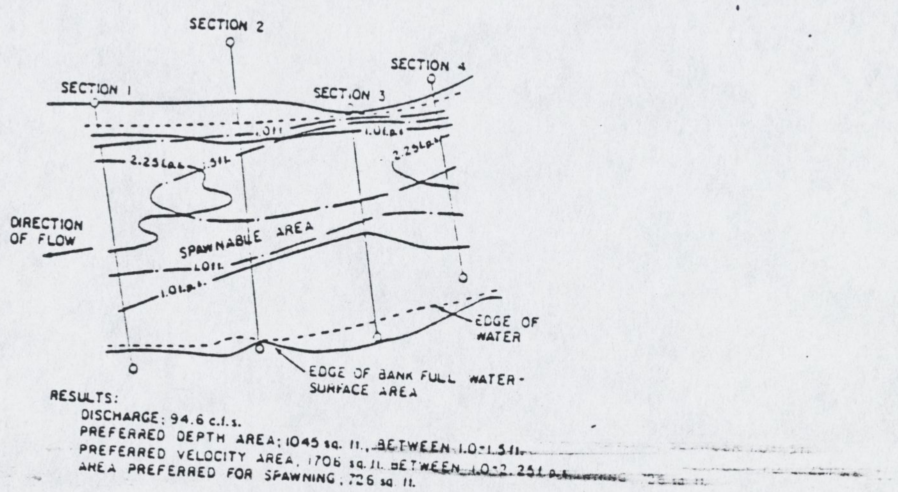


Figure 7b. Example of method for determining area of study reach that is preferred for spawning by fall chinook salmon at one river discharge, North Nemah River(from Collings, 1972).

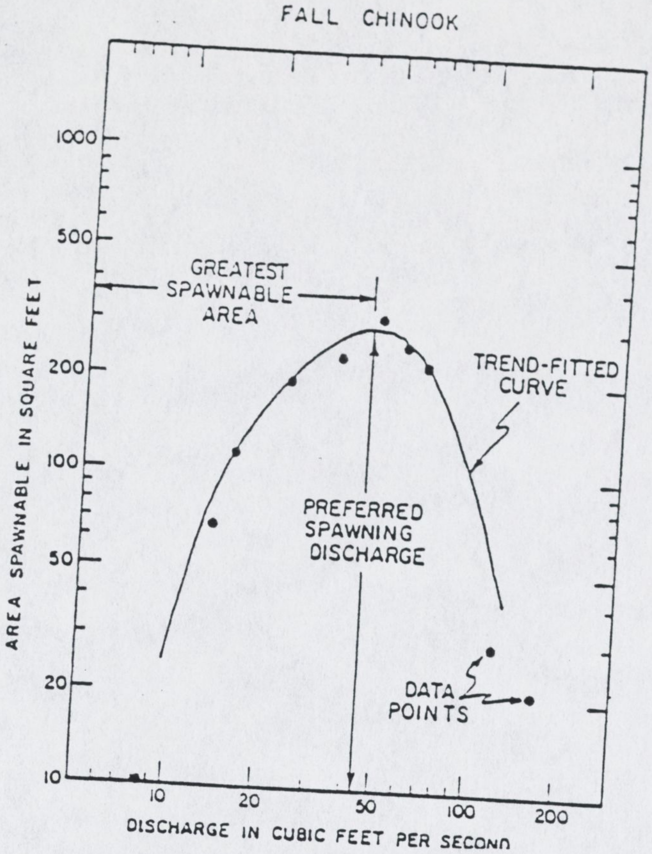


Figure 8. Method used to select the preferred spawning discharge, North Nemah River(from Collings, 1972).

The Usable Width Method was used to determine flow requirements for both migration and spawning of paddlefish. In areas where the spawning paddlefish were absent, the areas which provided the greatest sauger egg survival was used. In this case, the Habitat Mapping Method was used.

Bovee assumed that food was the limiting factor during the rearing stage. Since a pool's food base is continually replenished through drift from riffles, and riffles are more seriously affected by discharge reductions, riffle habitat was the limiting condition for rearing of fish. Therefore, he used a fast water species, the stonecat (Notorus flavus), as the rearing indicator. Again, he used the Habitat Mapping Method to determine the necessary flows.

The minimum spawning, migrating, and rearing flows were compared and whichever was greatest was used as the minimum recommended flow. One advantage of this method is that it is based on the biology of species which inhabit the stream. However, recent developments in modeling stream flows have antiquated the technique.

R-2 Cross Method

This method was the basis for an entire family of instream flow techniques, several of which are known by multiple names. They include the Single Cross-Section, Colorado, Critical Area, and Sag-Tape Methods.

The original procedure was developed by the U.S. Forest Service, Region-2 (Silvey, 1976). In this procedure, the entire river is broken up into study segments based on biological, hydrological, water quality or other parameters of concern, e.g., an important sports fishery or an area of great flow accretion. Single transects are then placed at critical or representative study sites. If critical sites are chosen, the investigator assumes that flows must be maintained at these critical sites to protect the fishery. If representative sites are selected, it is assumed that the transects will act as indicators for the entire stream segment. Usually the shallowest area of the shallowest riffle is used and considered a critical site for fishery considerations. All sites are marked and photographed.

Stakes are placed in the ground at both ends of the transects and leveled with each other using a string-line level or Abney level. A steel tape or chain with a known weight/foot is stretched from the top of one cross section stake to the other using a tape clamp and spring scale. Next, measurements from the tape to ground surface or channel bottom are taken to construct a bottom profile and velocities are recorded at intervals along the tape. A master reference point upstream is established and discharge (CFS) is taken to determine the stage.

These data are then inputted into the R-2 cross or similar computer program which calculates parameters listed in Figure 3. The program calculates the portion of the stream which meets or exceeds previously established depth requirements for the species of interest for fishery purposes. Predictions are repeated for various flows until the minimum flow which will support a target species is determined. Since all transects are tied into a single master reference point, only a single set of stream measurements must be taken.

IFG1

IFG1 is a modified version of the R2-Cross computer program and was developed by the Cooperative Instream Flow Group (recently reorganized as the Aquatics Systems Branch of the National Ecology Center) of the U.S. Fish and Wildlife Service (Bovee and Milhous, 1978; Hilgert 1982). The primary difference between the two is that IFG1 will also predict widths of streams having specific depths in addition to the parameters predicted by the original R2-Cross program.

The Hunter Creek Method (Boaze and Fifer, 1977) is a further refinement of the R-2 Cross Method. Several transects were selected for each study site, each representing at least one habitat type necessary for trout production, food production, cover, spawning areas and fish passage. Hydraulic measurements were made at each transect at seven different flows negating the need for computer simulation of flows, but greatly increasing the amount of fieldwork. Habitat criteria for water velocity, depth, and wetted perimeter were developed from a literature search and discussions with species experts. The hydraulic data were then used to calculate the value of each habitat variable at each of seven different flows. The subsequent recommended flow was the lowest of the seven flows which met at least two of the criteria as previously established.

Both Jesperson (1979, 1980) and Wesche et al. (1977) used similar techniques on many of the same streams in southwestern Wyoming. Essentially, it was the R2-Cross Method with multiple transects at each study site. The R-2 Cross computer program was used to simulate hydraulic characteristics at different stages.

Region 4 of the USFS developed a technique based on stream survey methods and cross-channel transects (Tew et. al., 1977). Five transects are placed across predetermined critical reaches at 50 foot intervals. The transect tape is tightly stretched across the stream and hydraulic measurements taken. At the same time a measure of flow is recorded and this is termed the index flow. All other flow simulations are related to this index flow. Average velocity, depth, width, area, and wetted perimeter are calculated and used as a base for comparing the amount of habitat at different flows.

A range of different flows are simulated using Manning's equation. The habitat available, i.e., hydraulic measurements, at the index flow is arbitrarily given a value of 100 percent and other a plot of these values versus discharge is made (Figure 9).

Weshe and Rechard (1980) state that the index flow measurements should be made during the lowest possible summer flows. Also, the R-2 Cross computer program can be substituted for the Manning's equation and used to calculate the predicted hydraulic parameters at additional flows. A major drawback of this technique is that the index flow is rather arbitrary. No site specific biological data were used to determine what the index flow should be.

Waters' Method

Waters (1976) presented a multiple transect method for determining optimum flow for four habitat parameters for several California trout species:

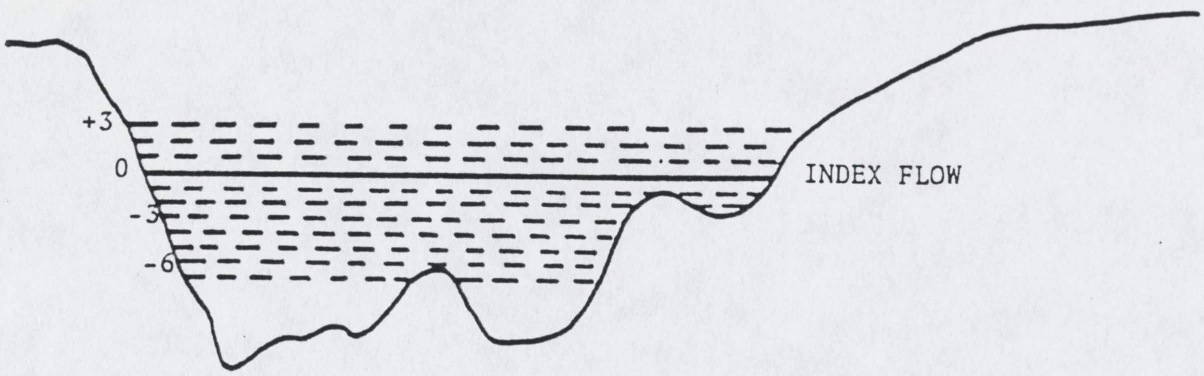


Figure 9. A channel profile showing the measured Index flow level and other selected flow levels.

resting microhabitat, food production, spawning, and an optional cover parameter.

This method starts by breaking the stream into study segments, then transects are placed across the stream which represent the typical habitats within a study section, however, Waters reasoned that major pools are less affected by streamflow reductions than other habitats, and therefore, are avoided. When transects are inplace in regulated streams, the highest streamflows to be measured are released and flows are allowed to stabilize. In non-regulated streams, the investigators must wait for different streamflows. Habitat parameters are measured, with velocity taken at a constant 0.2 feet above the substrate. Single digit substrate and cover codes which identifies the parameters at the point of measurement, are used and photographs are also taken.

The literature is reviewed and local experts consulted to determine the optimum value for each habitat parameter for each species at its different life stages. Once this optimum value is determined, it is given a relative value of 1.00 and other possible values for this parameter are rated based on this optimal value. Transect data are then entered into a computer for analysis. The relative values at each measured point, for each parameter, e.g., depth, velocity, cover and substrate, at each flow are calculated. From these data the computer produces data and/or plots of the relative values to fish species for each of the four habitat parameters measured. The computer output consists of the following: 1) Total relative units for each series of transects within each station, including a station total; 2) Total relative units for each series of stations within the stream section under study, including a stream section total; 3) Mean relative units of lands; 4) Standard deviation of values in 1 and 2; 5) 90 percent confidence intervals for 1 and 2; and 6) Relative distribution of different substrates is also given. This information is then used by biologists to determine the optimum flow for the species in question at different life stages. The program can also multiply the mean relative units by the actual streambed area to get a measure of quality of habitat per square foot or meter for each habitat parameter. This can be used to compare values from one stream to another.

This method adds several new features to the R2 Cross Method family, specifically multiple transects and mean habitat values per unit area of stream.

Instream Flow Incremental Methodology

Instream Flow Incremental Methodology (IFIM) is the standard methodology used by the FWS and was developed by the Cooperative Instream Flow Service Group in Fort Collins, Colorado. It provides information on the effects of a variety of flow regimes, and as such, can be used for negotiations.

There are several aspects to the methodology. Biological criteria need to be developed for the species of interest. Biological criteria were developed for several of the previous methods, but it is a cornerstone of this method and emphasis is placed on collecting biological data from the stream to be modeled. Biological data are incorporated into an electivity curve, a two dimensional plot representing the relative suitability of a variable (D, V, S, or cover (C)) for a fish life stage (Figure 10). These curves which are based

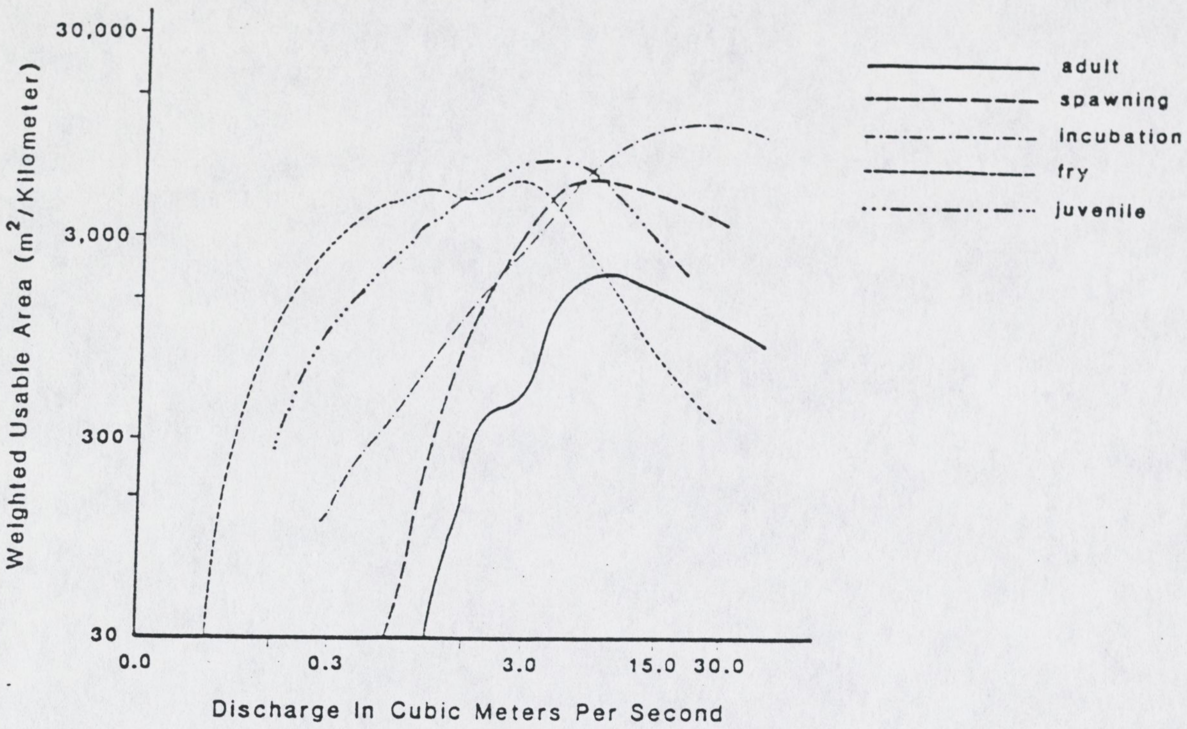


Figure 10. Habitat/discharge relationship for a rainbow trout in a sample reach (from Milhous, 1984)

on field data may be either utilization (type 2) curves, or preference (type 3) curves. The former are based on where the species is most likely to be found under prevailing conditions and the latter are an estimation of where the organism would be found were all possible conditions available. A separate curve is created for each life stage of each target species; adult, juvenile, larval, and spawning. For a more complete explanation of how these curves are created and used, please refer to Nelson, 1984.

Another major aspect of IFIM is its hydraulic simulation capabilities. It will predict the values of hydraulic parameters for a range of flows.

In the past, several different routines were used for the hydraulic simulation depending on the physical structure of the stream in question. It is now recommended that a single set of IFG4 measurements be used to calculate the velocity distribution across a channel. A stage-discharge relationship is then established using one of three methods: a rating curve, a step-back order simulation or normal depth model.

The ~~rating curve~~ stage discharge relationship is determined in one of three ways. First, the log-log linear relationship between water surface elevation (WSL) and velocity may be determined. This is done with at least three sets of discharge and WSL measurements, i.e., the traditional IFG4 procedure. Second, a step-back order simulation uses Bernoulli's and Manning's equations to establish the stage-discharge relationship, e.g., Water Surface Profile (WSP) or Hydraulic Engineering Center (HEC) computer programs. Lastly, a normal depth model is produced using MANSQ or R2 Cross computer programs.

These are all recent innovations in IFIM and have not been widely published. It goes under the name of the combined method of IFG4A. When using any of these simulations, it is important that the investigators calibrating the model understand the theory behind the simulations.

IFIM can incorporate ~~r~~seasonal flows, and flow recommendations can be made which take into account both the seasonal flows and requirements of the different species' life stages. This can be accomplished by developing a habitat time series. Monthly mean or median flows are entered into a computer program and habitat values (WUA) are given for each species' life stage, for each month and normal or baseline conditions are established. Goals can then be established, for example, determining a flow which retains the median amount of WUA for spawning during the spawning seasons and this same amount of adult habitat during the rest of the year for rainbow trout, and flows which meet these goals determined.

Recommendations

After looking at the existing techniques available to quantify instream flows we find that we cannot recommend a single method as the best. Instead, the techniques used should be determined by the level of accuracy which is needed. If flow recommendations are likely to be challenged in court or need to be precise it may be prudent to select one of the hydraulic methods, if court challenge is not imminent or flow needs are general then a hydrographic method may be method of choice.

We evaluated each method within both groups of techniques, hydrographic and hydraulic, and tried to determine what methods in each group would work best for Arizona streams. The hydrographic techniques were evaluated by using stream flow data from two United States Geological Survey (USGS) gaging stations on the Verde River: USGS gage number 5037, Verde River near Paulden and USGS gage number 5085, Verde River below Tangle Creek. These two gages are good choices because, while they are on the same river, they represent different flow regions. The Paulden Gage is located in the upper reaches of the watershed and is characterized by a fairly uniform flow where the base flow and median flow are approximately equal (Figure 1). The Tangle Creek Gage on the other hand, shows flows which are perhaps more typical for desert streams, i.e., one with sharp season swings (Figure 2). Data from these two gages will serve to cover a broad spectrum of flow conditions. Figures 1 and 2 show the results of the comparisons of hydrographic techniques and it is obvious that a "first cut" of techniques can be readily made. All the Tennent Methods fail to make the cut because they do not recommend flows which match up well with existing conditions at both gaged locations. The remaining two methods, the South Dakota Method and Great Plains Method offer promise because they take into account flows on a monthly basis. The South Dakota Method would give generous flow recommendations relative to the median flow at the Tangle Creek location, however, it would allow flows to dip up to 30 percent below the median at the Paulden Gage location. The Verde River at the Paulden Gage is a small, headwater stream where flow reductions of this magnitude would be undesirable. Recommendations based on the Great Plains Method allows less flow at the Tangle Creek location than the South Dakota Method, however, it does parallel the median flow at the Verde River at Paulden quite closely. The Great Plains Method is more closely tied to monthly flows than any of the other methods and, because of this, will better mimic the existing hydrographic that the other hydrographic methods reviewed. Because of its close tie to monthly (seasonal) flows we feel it is the best of the methods we evaluated and therefore, recommend its use. This recommendation is based on the assumption that the best methods is the one which mirrors existing flow conditions.

The hydraulic group of techniques is more difficult to evaluate. This is because data must be obtained on-site and is not available in existing publications, however, the evaluation process is made a little easier by the fact that these techniques have envolved over a period of time and several have become outdated. This has been due the increased use of the computers to process data. The methods which we reviewed and feel are either not applicable to Arizona or antiquated are as follows: Hoppe and Finnel Method, New England or Connecticut Method, Idaho Method, Habitat Mapping, Indicator Species Overriding Consideration Method, and Waters' Method. The hydraulic methods which remain are: The Wetted Perimeter Method, Useable Width Method, R-2 Cross Method, IFG1 and IFG4A. The Wetted Perimeter Method is the most basic and easiest of the hydraulic methods to use. We recommend this method if a critical habitat, normally a riffle, can be defined as indeed critical. The Useable Width and R-2 Cross methods are both acceptable.

IFIM is the state of the art method for instream flow quantification. If flow recommendation are likely to be challenged in court or precision is needed, we recommend IFG4A, but we caution that the investigators should become familiar and comfortable with the limitations and manipulation of the methods used to determine the stage-discharge relationship.

At present, none of the existing techniques take flushing flows into account, although the Fish and Wildlife Service is researching this problem using IFIM. These flows may be extremely important to maintain Southwestern systems and future evaluations should consider this.

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