A GUIDE FOR THE SELECTION OF STANDARD METHODS FOR QUANTIFYING SPORTFISH HABITAT CAPABILITY AND SUITABILITY IN STREAMS AND LAKES OF BRITISH COLUMBIA

Submitted to

B.C. Environment, Fisheries Branch
Research and Development Section
Vancouver, B.C.

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Preamble

The Resources Inventory Committee consists of representatives from various ministries and agencies of the Canadian and the British Columbia governments. First Nations peoples are represented in the Committee. RIC objectives are to develop a common set of standards and procedures for the provincial resources inventories, as recommended by the Forest Resources Commission in its report The Future of Our Forests.

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EXECUTIVE SUMMARY

The protection and management of fish habitat is presently considered a priority for integrated resource management within provincial and federal agencies. Habitat capability and suitability models are needed to assist in management decisions, explore relative benefits of mitigation techniques, evaluate new stream and lake restoration methods, and assist in evaluation of habitat impacts. The purpose of this guide is to identify the most appropriate capability and suitability models for use in B.C., review their limitations and benefits, and provide recommendations on model application, model validation, and future analysis.

We reviewed 91 stream, and 87 lake capability models and synthesized information which could be used to evaluate the predictive abilities of each. Within lake and stream categories, each model was ranked according to a single statistical indicator, the product of the coefficient of determination and degrees of freedom. For the top-ranked models, a more detailed evaluation was used to determine the most likely candidates for use in the province. This model selection process was repeated within each of the 4 model use categories (stock management, recreational and regional planning, habitat impacts and mitigation, habitat restoration and improvement). The top-ranked models fell into one or more of 3 spatial scales defined by the Resource Inventory Committee:

1) *Overview Level* for regional or sub-regional applications used to identify where to manage rather than how to manage.

2) *Reconnaissance Level* for local/basin applications used in the classification and management of groups with similar features.

3) *Intensive Level* for operational applications used in the management of individual stocks.

The top-ranked models at the overview and reconnaissance levels were considered potentially useful as interim methods, however validation and additional model analyses using B.C. data were strongly recommended. The intensive level capability models reviewed were considered inadequate because they have not been validated in B.C., yet the management level requires precise and accurate predictions. It was recommended that experimental manipulations be used to evaluate habitat impacts and mitigative measures for high-value resources.

Process-oriented models may include empirical relationships relating populations to habitat capability or suitability, together with a representation of population dynamics. Process models fill 3 different niches with respect to fish habitat issues: 1) to evaluate fisheries policies in relation to a specific site's capability; 2) to estimate the impact of watershed disturbances and the potential benefit of mitigative measures; or 3) to improve understanding of ecological processes. We have provided a review of different process models in each of these 3 categories as examples of these different approaches. We have also provided recommendations concerning the development, refinement, and analysis of future process models to be used within the B.C. Ministry of Environment, Lands, and Parks.

Six suitability methods were evaluated for application in B.C. These are the Habitat Evaluation Procedure (HEP), the Instream Flow Incremental Methodology (IFIM), the Missouri Stream Habitat Evaluation Procedure (SHEP), the Fish Habitat Index (FHI), the Planned Reservoir Habitat Suitability Index, and the Tennant Flow Index. The Habitat Evaluation Procedure (HEP) has been tested at a few sites in British Columbia with poor results due to inappropriate selection of habitat variables and a general lack of site specific SI curves. The biggest problem with HEP is that it requires extensive *a priori* knowledge of factors that limit fish abundance at a given site. At the intensive management level, the Instream Flow Increment Methodology (IFIM) can be immediately used in the province, but only for large, big budget projects requiring an estimate of the change in suitability with respect to a flow manipulation. The Tennant Flow Index has been picked up as a quick approach for estimating suitability in various regions in B.C. However, it has never been formally validated and this testing must be completed before it can be recommended for routine use. Each of SHEP, FHI, and the planned reservoir HSI were developed primarily as concepts that the authors have indicated require further research and testing before they can be applied elsewhere with confidence. These methods have never been applied to sites in British Columbia and as part of a long term strategy for methods development, they should be considered for additional testing. However, they do have problems other than the lack of validation that must also be considered as part of any testing initiative.

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1.0 INTRODUCTION

The protection and management of fish habitat is considered a priority for integrated resource management within Provincial and Federal policy documents. In British Columbia, the Aquatic Inventory Task Force (AITF) of the Resources Inventory Committee (RIC) is currently reviewing all aspects of aquatic inventory data collection and interpretation in order to establish standardized methods to assess habitat suitability (a measure of habitat quality usually expressed in terms of an index or relative value) and capability (measure of carrying capacity, density, number of fish, biomass, etc.) for freshwater sportfish in B.C. The AITF recognizes that approaches to assess habitat suitability and capability are controversial. Practical applications of the Instream Flow Incremental Methodology (Bovee 1982) have been criticized regarding violation of assumptions (Mathur *et al.*1985, Shirvell 1986). These criticisms have been countered (Orth and Maughan 1986), and the debate continues. A general finding that habitat models are poor predictors of habitat capability when applied to sites other than where they were developed (Shirvell 1989) has raised concern over the liberal use of models as predictive tools in fisheries management (Bisson 1992). However, habitat capability and suitability models are needed to assist in management decisions, explore relative benefits of mitigation techniques, evaluate new stream and lake restoration techniques, assist in the evaluation of impacts, help in providing a focus for research needs, and contribute to habitat protection planning. Faced with these demands, fisheries managers cannot ignore modelling tools. Instead there is a need to actively contribute to the development or improvement of assessment techniques. Critical review of existing approaches is essential, but when it uncovers problems the inevitable statement in the decision making process is, "I know the model has problems but show me something better." That comment clearly instills the fact that habitat models will continue to be used and effort must go towards improvement, not outright rejection.

In the process of developing management tools, the AITF has identified the need for standard methods and interpretations that can be applied at the *overview*, *reconnaissance*, and *intensive* levels of management in anadromous rivers, inland rivers, and lake habitats. The three management levels are defined by RIC as follows (Anon. 1992):

> *Overview Level* is for regional or sub-regional applications used to identify where to manage rather than how to manage.
>
> *Reconnaissance Level* is for local/basin scale applications used for classification and management of groups of similar features.
>
> *Intensive Level* is for operational applications used for the management of individual stocks.

To begin the process of developing standard methods, a bibliography of capability and suitability methods was prepared (Aquatic Resources Ltd. 1993). With that information compiled, an evaluation was required to select the most appropriate habitat methodologies for use in British Columbia which in turn, could be used to focus RIC data collection efforts.

In this report, we describe the results of a process to evaluate habitat capability and suitability models. In Section 2.0, we describe the set of quantitative and qualitative criteria used to select the most appropriate habitat capability models or approaches for use in B.C. For the top-ranked capability models, we discuss the benefits and limitations of each approach, and compared the model data requirements with the information currently available for B.C. We provide recommendations and cautions for applying the top-ranked models and for the development of new models based on B.C. data. In Section 3.0, we provided an overview of process models which can be used to estimate habitat capability and discuss the advantages and drawbacks of process *vs*. empirical approaches. In Section 4.0, we review different habitat suitability methods, and describe their limitations and key assumptions.

A decision tree, structured similarly to this report, can be used to select the most appropriate methodologies for different stock management/habitat management situations (Fig. 1.0). Habitat capability methods can be selected based on 4 possible *use categories*:

1. Stock Management, where measured fish population status is compared to estimates of predictive capability;

2. Recreational/Regional Planning, where coarse estimates of stock capacity are estimated from map-based or easily obtained information;

3. Habitat Impacts and Mitigation, where predictions of impacts of logging and flow reductions are required; and

4. Habitat Restoration and Improvement, where predictions of benefits from managed changes in habitat complexity (restoring large woody debris in historically logged or channelized streams, adding boulder clusters, and stream fertilization) are required.

Process models developed within the B.C. Fisheries Branch may be employed by those with a thorough understanding of the models for a detailed evaluation of fisheries regulatory policies at the intensive management level. Habitat suitability methods fall into 2 basic categories. Instream Flow Incremental Methodology is the recommended approach for detailed investigations of large projects with sufficient resources (in the $100,000's) for the impact assessment. Other habitat suitability assessments can use a variety of simpler approaches which are summarized in Section 4.0.

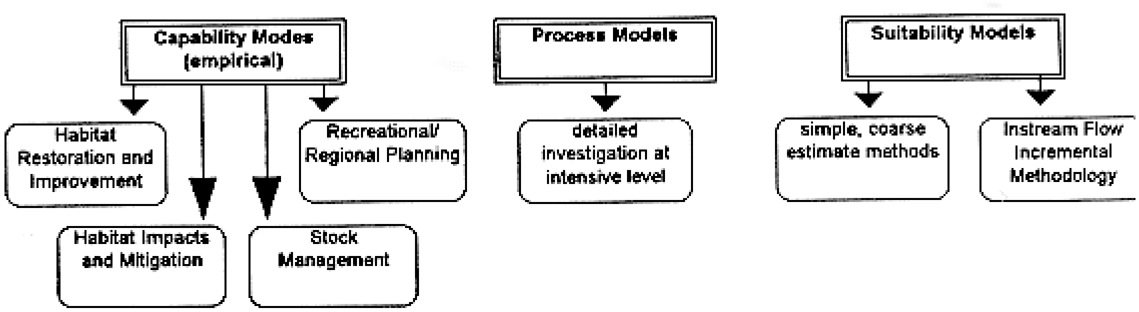

Figure 1.1 Decision tree for selecting alternate approaches for habitat management and stock assessment. See text for details.

2.0 SELECTION OF EMPIRICAL CAPABILITY MODELS

Prediction of fish yield or biomass in aquatic environments has long been a goal of fisheries managers as evidenced by the great number of capability models in the literature. In this section of our analysis, we have sorted through a vast number of empirical models which have been developed to predict habitat capability, and selected the most suitable ones for use in B.C. Many of the empirical models we evaluated were intended only to describe the data the investigators measured, and the danger of applying empirical models to conditions other than the ones under which they were developed (*e.g.*, different geographic locations) is well recognized. Despite the danger or uncertainty of applying models to other systems, an evaluation of their relative merits is nevertheless useful and required for two reasons. First, existing empirical models can provide a *rough index* of habitat capability to resource managers who have no other estimates of capability. Second, and more important, an evaluation of existing models will identify which variables and approaches have been the most successful at predicting habitat capability in other areas. These variables and approaches can then be used to guide: 1) analysts in developing capability models specific to B.C. and 2) the Aquatic Task Force of the Resource Inventory Committee (RIC) in future data collection initiatives.

This section of the report is divided into five parts. We first discuss key assumptions implicit in the use of empirical models to predict habitat capability. Second, we provide a brief review of the statistical measures used to evaluate the predictive abilities of capability models. Third, we describe our methods for qualitatively evaluating the capability models. Fourth, we describe the results of our evaluation, and provide brief reviews of the top-ranked models. Finally, we provide some general recommendations and cautions for using these models and ideas for developing similar predictive relationships using B.C. data.

2.1 Issues in Interpreting Models for Estimating Habitat Capability

Population responses to changes in habitat capability can manifest themselves as changes in:

1. *carrying capacity,* the maximum population size (maximum density or biomass) that can be sustained in a habitat over the long term; and/or
2. the *intrinsic growth rate of the population*, the doubling time at very low population densities (affected by survival, growth, and fecundity).

When using models to predict habitat capability it is very important to distinguish whether the model predicts true carrying capacity or the expected population size at a given recruitment (which is determined by the intrinsic growth rate and carrying capacity). A model which predicts egg to fry survival as a function of sediment levels increased by logging, for example, quantifies changes in the intrinsic population growth rate. Increase sediment levels reduces survival rates, but does not change the carrying capacity of the habitat affected (Figure 2.1a). It quantifies the change in the number of fish produced for a given egg deposition. In such a situation, carrying capacity is a not a good indicator of the influence of the perturbation on habitat capability. On the other end of the spectrum, if logging leads to a reduction in the amount of large organic debris in a stream and reduces habitat complexity, the system may still be able to support the same number of fish at very low egg deposition rates relative to an unlogged watershed, yet the true carrying capacity of the system is reduced (Figure 2.1b). In this case, a model which predicts changes in carrying capacity is an effective tool for predicting the impact of logging on habitat capability. In this example, both models predicted a change in habitat capability resulting from an impact, but tracked different population responses to that change.

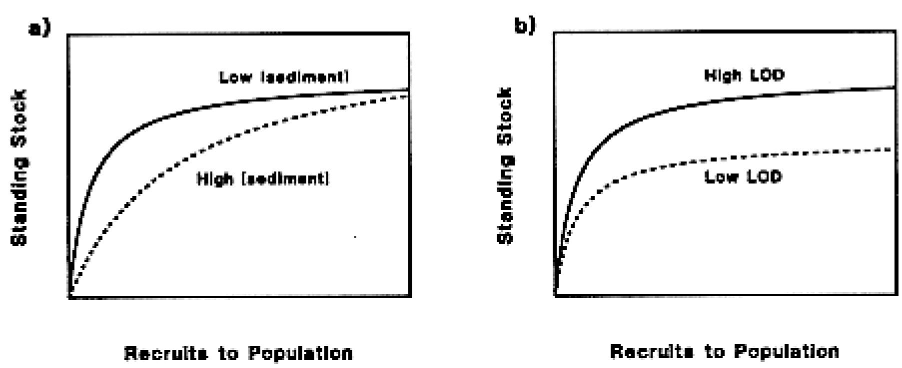

Figure 2.1 Theoretical relationships between abundance (*e.g.*, standing stock) and recruitment to the population in stream habitats affected by logging In a)

reduced egg-fry survival resulting from sedimentation has no effect on carrying capacity, but does reduce the intrinsic rate of population increase. In b) decreases in large organic debris (LOD) reduces the carrying capacity of the population.

A second set of issues to keep in mind when evaluating capability models is that the vast majority of models are based on observational, rather than experimental data. The lack of control across systems on processes such as juvenile recruitment, fishing mortality, catchability, interspecific competition, or predation introduce considerable noise into the models and affects their predictive abilities. In spite of the limitations and problems of empirical predictive models to assess habitat capability they have been extensively employed in freshwater fisheries management because they are easy to use and can provide coarse estimates of abundance. Potential users of these models should be aware however, that the predictions may have little to do with the absolute carrying capacity, and may be biased due to uncontrolled factors.

A third set of issues is unique to capability models based on harvest indicators (*e.g.*, catch per unit effort, maximum sustainable yield). These types of models can be biased by differences in fishing mortality rates among systems. Figure 2.2 depicts a typical stock-recruit (S-R) curve, in this instance, specified as a Ricker model. Catch is the height of the Ricker function above the replacement line. MSY is the maximum distance between the S-R curve and the replacement line. The spawning stock biomass at catch=0 where the S-R function crosses the replacement line, is the unfished equilibrium or carrying capacity of the system. From this figure, it is clear that harvest-based indicators do not represent the carrying capacity of the system (i.e. the unfished equilibrium). Since catch can be produced through a range of spawning stock densities in a single system (as determined by the fishing mortality rate), one does not know how close the catch indicator is to the carrying capacity. This problem can introduce considerable bias into harvest-based models where sites with different fishing mortalities are used to generate the predictive relationship. Notwithstanding these problems, models between harvest and easy-to-measure variables are abundant in the literature, principally because of the availability of catch data. Harvest-based regression models generally have larger sample sizes than non-harvest models and to some extent this may offset the increased variability associated with harvest-based indicators for tracking capability.

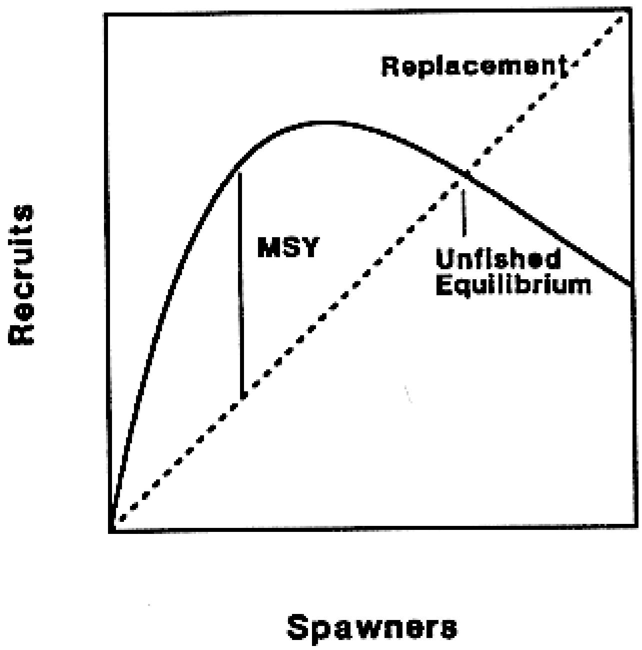

Figure 2.2 A Ricker stock-recruitment function showing the spawner biomass for maximum sustainable yield (MSY). The unfished equilibrium, or carrying capacity, is the point where the recruitment function crosses the population replacement line.

2.2 Statistical Issues for Evaluating Predictive Capabilities of Regression Models

When evaluating a regression model two questions naturally arise:

1. Does the model fit the data adequately?
2. Will the model predict responses (either through interpolation or extrapolation) adequately?

A number of different regression statistics can be used to evaluate the model fit and its predictive abilities. These include the coefficient of determination (the proportion of explained variance in the dependent variable), prediction variance, and other indicators such as confidence limits. The probability of the regression slope being significantly different from 0 is the P value commonly reported with regression output. The rejection of the null hypothesis means that a statistically significant trend is detected, however nothing is implied concerning the quality of fit of the regression line or its ability to predict (Myers 1986). P and R^2 are related since the lower the P value, the greater the ratio of explained variance to residual variance. The prediction capabilities of a regression model are influenced by the sample size and spread of the data range of the regressor variable. Prediction capabilities are improved by an increase in the sample size (assuming all other things are equal) and when the input value of the regressor variable is close to the average of the regressor values (Myers 1986).

It is important to make the distinction between fitted values from a regression model and prediction. Prediction applies to regressor values where interpolation or extrapolation is necessary, i.e., where a value of the regressor variable is not contained in the data set used to derive the model. Statistics such as the standard error of the predicted values or confidence limits do give some indication of the model for interpolation but say nothing about the model's performance in the area of extrapolation (Myers 1986).

2.3 Methods for Model Evaluation

2.3.1 Literature Collection

In 1993, Aquatic Resources Limited completed a project commissioned by the Aquatic Task Force of the Resource Inventory Committee to assemble, organize, and summarize existing methods for assessing habitat capability and suitability methods for freshwater sportfish in British Columbia. This report, "Sportfish Habitat Suitability and Capability Literature Review" (Aquatic Resources Ltd. 1993), accompanying bibliography and collection of references was used as a basis for our model review. In total, we reviewed 119 papers in our search for the best habitat capability methods. These papers were collected from three sources:

1) Aquatic Resources Ltd. (1993);
2) references cited in the Aquatic Resources Ltd. bibliography, but not compiled; and
3) recent references (as of September 1993) catalogued on the BIOSIS database using the search strategy reported by Aquatic Resources Ltd. 1993.

2.3.2 Literature Classification and Model Cataloguing

Each paper was briefly reviewed to select those which contained empirical capability models. The significant regressions in each paper were first classified in an EXCEL database according to habitat and one of the species groupings listed below. This list is a subset of highly-valued freshwater sportfish managed by the Province agreed upon during the first project scoping meeting on November 23, 1993. These are:

1. Trout	rainbow trout (RBT), steelhead trout (SHT), brown trout (BNT), cutthroat trout (CT), and those species reported as "trout" (T);
2. Char	lake trout (LT), brook trout (BKT), and bulltrout/Dolly Varden (Bull/DV);
3. Salmon	pink (PS), coho, chinook (CHIN), and those species reported as "salmonids" (S);
4. Other	. Other northern pike (NP), walleye (wall), whitefish (WF), arctic grayling (AG), white sturgeon (WS), and kokanee (KOK);
5. Mixed	a combination of 2 or more of the above species groups, or those models which reported the dependent variable as mixed species.

Information describing the dependent variable, independent variables, the area for which the model was developed, related papers, and whether the model was validated was also summarized in the database. A printout of the database is provided in Appendix A.

The capability models were preliminarily ranked using statistical descriptors to assess each model's prediction abilities. Our approach was similar to that of Fausch *et al.* (1988) who reviewed 99 stream habitat models. Although we compiled a variety of statistical descriptors for each model reviewed (*e.g.*, N, P, EMS, SE, R^2) we found that only 3 descriptors were consistently reported. Thus our preliminary ranking process was limited the three statistical criteria: sample size (N); the number of terms used in the model; and the coefficient of determination or multiple determination (r^2 or R^2, referred to hereafter as R^2). Papers which contained capability models but did not report these 3 statistics used in the ranking process (*i.e.*, they instead reported statistics such as confidence limits, standard error of slope, standard error of prediction, or coefficient of variation) were reclassified and not included in the analysis.

To incorporate the ranking criteria R^2, N, and the number of terms into a single index, we multiplied the coefficient of determination by the degrees of freedom (DF = N minus the number of terms) remaining after the model was specified (residual DF). R^2 describes how well the data fit the model. Degrees of freedom is an index of sample size adjusted for the number of terms used in each model. Our composite indicator R^2*DF has no theoretical basis, however it is appealing because it weights explained variance and degrees of freedom equally. This estimator provides a conservative means of assessing the "transportability" of existing models to other systems because generally, as sample size increases, the range of the regressor variable also increases. Thus when applying the model in other systems, it is more likely that independent variables fall within the data range used to derive the model coefficients. In other words, we assumed that larger data sets increase the chances that the model will be used to interpolate, rather than extrapolate estimates of the dependent variable. A larger sample size will also reduce the chances of bias in model coefficients resulting from a few anomalous data points providing that the distribution of the regressor variables meets the assumptions of least squares regression.

Within the stream and lake habitat-type categories, the models were sorted by the composite indicator R^2*DF. We then evaluated the top-ranked lake and stream models based on qualitative criteria, essentially looking for reasons to exclude each model. These qualitative exclusion criteria fell into four general categories:

- Large differences between the geographic location where the model was developed and B.C. For example, predictive models based on reservoirs in the southeastern U.S. were not considered transferrable to B.C. systems due to differences in the length of the growing season and species assemblages.
- Large differences in the range of the regressor variables relative to the range of the variable in B.C. For example, we excluded a model which predicted lake harvest based on elevation developed from a set of lakes in Ontario. Clearly, the range in altitude of B.C. lakes would exceed the range in the regressor variable from the Ontario-based relationship.
- Limited potential utility of the model For example, models that predict lake harvest using only effort or fish weight were excluded, since catches would generally be known for systems where effort or fish size data were available.
- Poor application of statistical methods For example, we excluded stream habitat models which used many variables when no assessment or correction of potential collinearity problems was made.

Our reasons for rejecting top-ranked models were recorded in the capability model database (Appendix A).

To determine if the data requirements of the model were compatible with the present biophysical inventory of B.C., we compared the input variables of each model to the environmental parameters catalogued in existing databases and information sources. Information on existing databases (*e.g.*, Fish Information Summary System, B.C. Lakes Database, SEAM) and other sources (paper files and future developments) was provided primarily by Dave Tredger (Fish and Wildlife Branch, MOE, Victoria). This comparison was done to assess the ease of using the model in terms of currently available data, not to rank the model. More importantly however, this comparison was used to highlight inventory needs for RIC planning purposes. Data availability information (classified as yes, no, unknown) is included in the database tables in Appendix A. The reader should be aware that this was a "loose classification" and was in many cases simply Dave Tredger's best estimate of what has already been collected for at least some systems and years.

We classified the top-ranked models into one or more of the 3 management levels *overview, reconnaissance, and intensive* defined by RIC (Anon. 1992). Because the classification of each model was based on the data requirements and spatial scale of the model, there can be some overlap in the classification. According to RIC definitions (Anon. 1992), overview level models are those which can be applied on a province-wide basis and require data that are primarily office-generated or exist in electronic databases, most often derived from maps, small-scale photographic imagery, and lake and stream surveys. Overview models identify where to manage rather than how to manage. Reconnaissance models need to be widely applicable to high value/high potential impact areas. They have greater data requirements (than overview models) which are met through relatively inexpensive field programs. These models are used at the local/basin scale for classification and management of groups of similar features. Intensive models are used for high priority areas at specific sites for the management of individual stocks. These models generally use data based on extensive field work where biological and physical variables are measured in detail.

2.4 Results of Model Evaluation

In total, 87 lake and 91 stream capability models applicable to highly-valued freshwater B.C. sportfish were catalogued. This total includes 40 stream capability models which were reviewed by Fausch *et al.* (1988). Table 2.1 summarizes these models by capability models by habitat, dependent variable, and species group.

Table 2.1. Summary of empirical capability models by habitat, dependent variable, and species category

Habitat	Dependent Variable	Species					Total
		trout	char	salmon	other	mixed	
Lake	*harvest*	5	14	1	15	41	76
	production					8	8
	standing crop	1				2	3
TOTAL LAKE		6	14	1	15	51	87
Stream	*harvest*			2			2
	standing crop	42	9	20		18	89
TOTAL STREAM		42	9	22	0	18	91
TOTAL LAKE + STREAM		48	23	23	15	69	178

To describe the collection of models in our database using the statistical measures in our composite ranking indicator (R^2*DF), we plotted the relationship between the coefficient of determination and the degrees of freedom for all lake and stream models (Figure 2.3). In both habitat types it is clear that models with the highest R^2 have relatively small sample sizes. Clearly, the best models from a predictive standpoint would have a high R^2 and large sample size and would therefore be located in the upper right quadrants of the figures. We have also categorized the data points on the graphs by comparing the model input requirements with the present biophysical inventory of B.C. Only 16% of the lake capability models require data that are not available in one of the provincial data sources. Up to 39% of the stream capability models require data which may not be in one of the provincial data sources (18% data not available, 20% unknown data requirements, 62% data available). This is to be expected since the stream models generally have many more terms relative to the lake models (stream avg.=3.6 terms, max.= 21; lake avg.=3.2 terms, max.=6). Examination of the cumulative frequency distribution of the R^2*DF indictor by habitat-type shows that there is a clear break between the top-ranked models and the majority of other models (Figure 2.4). Less than 5% of the models within the lake or stream habitat categories had R^2*DF values greater than 60. Our method has separated the top-ranked models quite distinctly.

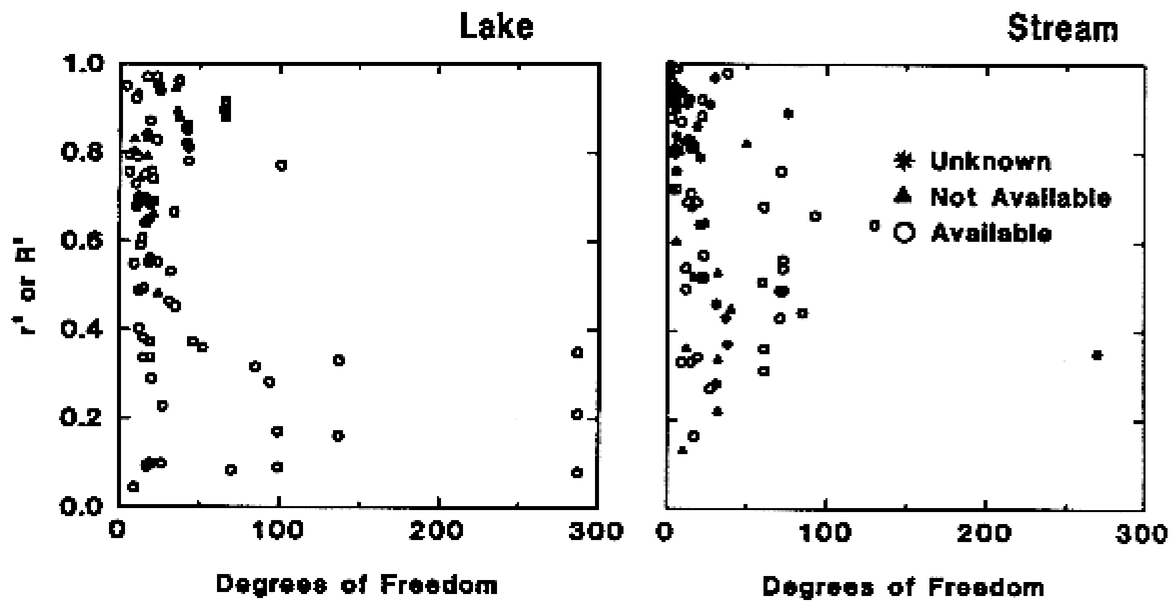

Figure 2.3 The relationship between the coefficient of determination and degrees of freedom in lake (a) and stream (b) capability models reviewed Data are segregated according to availability in existing provincial databases. To increase resolution for the majority of data points, we do not show extreme values for the stream models (max DF=1588).

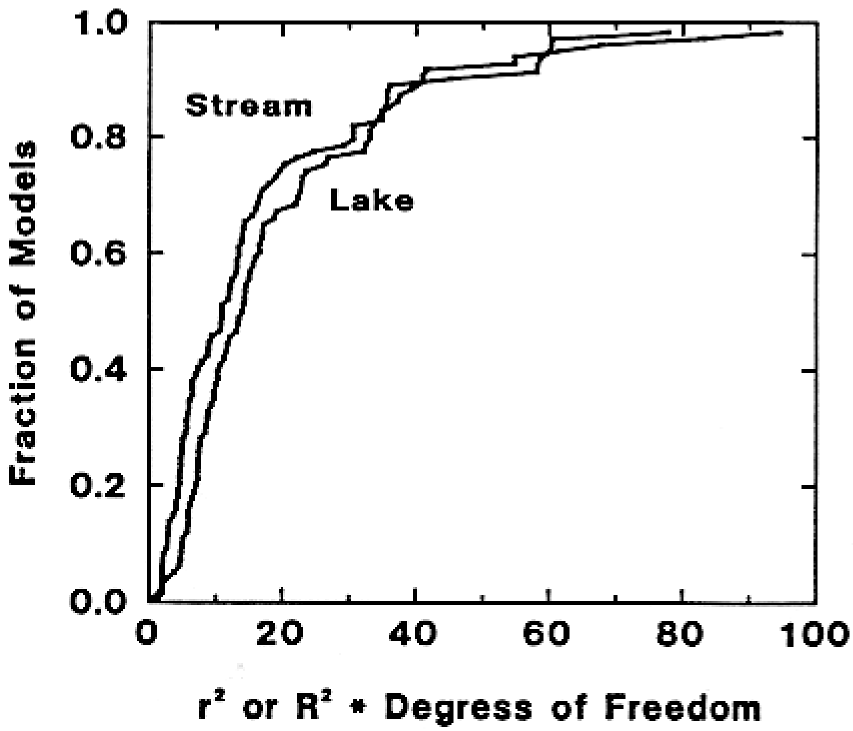

Figure 2.4 Cumulative frequency distributions of the composite ranking indicator R^2*degrees of freedom by lakes and streams. To increase resolution for the majority of data points, we did not show extreme values (maximum R^2*DF for lakes = 100.5, maximum R^2*DF for streams =1328)

In the next two sections we summarize the details of the top-ranked models in terms of their predictive ability, their applicability at different management levels (overview, reconnaissance, intensive), what they can be used to predict, general benefits of the approaches, and their limitations. For the top-ranked models which are described below, we have highlighted their limitations and recommendations for use in *italics* for emphasis. The detail of our review varies considerably between papers and reflects the amount of detail and discussion provided by the authors in their publications.

2.4.1 Stream Capability Models

Our review of stream capability models was conducted in two stages. First, we ranked the models in an overall sense, that is, independent of the intended use of the models. Second, we ranked the models within the 4 main *use categories*: These are:

1) *Stock Management*, where measured fish population status is compared to estimates of predictive capability;
2) *Recreational/Regional Planning*, where coarse estimates of stock capacity are estimated from map-based or easily obtained information for planning purposes;
3) *Habitat Impacts and Mitigation*, where predictions of impacts of logging and flow reductions are required; and
4) *Habitat Restoration and Improvement*, where predictions of benefits from managed changes in habitat complexity (restoring large woody debris in historically logged or channelized streams, adding boulder clusters, stream fertilization) are required.

2.4.1.1 Overall Model Ranking

For our overall ranking, we selected seven stream habitat capability papers containing a number of models. The models in these papers cover a broad spectrum of approaches, from remote techniques which are highly useful at an overview level, to very intensive methods requiring detailed information on flow, temperature, channel morphology, and cover characteristics. All predict standing crop and it is believed that the input data are available for all models (except 114) for at least some systems and years. The model descriptions in this section are presented in the order that the papers first appear in the table summary below.

Table 2.2. Summary of the overall top-ranked stream capability models. Management level: O=Overview, R=Reconnaissance, I=Intensive. Model use categories: SM=Stock Management; RP=Recreational/Regional Planning; HI=Habitat Impacts and Mitigation; HR= Habitat Restoration and Improvement. See Section 2.3.2 for species abbreviations.

Author	Mod. #	Spp.	Independent Variables				R^2	N	R^2DF	Mod. Use Cat.	Man. Lev.	Area Developed
			1	2	3	4						
Ptolemy *et al.* 1991	162.2	S	fish wgt.	alkalinity	fixed non-filter. residue		0.84	1592	1327.6	SM	R	diverse eco-regions of B.C.
Fraley & Graham 1981	69	CT, BULL	overhead cover	instream cover	stream order	substrate	0.64	134	83.2	HR	R,I	Flathead R. drainage (Mont.)
Sekulich 1980	114	CHIN	3 variables (not specified) are some combination of number of eggs deposited + channel morphometry, flow, temperature, and/or biological variables				0.89	80	67.6	HI	I	Idaho
Oswood & Barber 1982	105.1 (61.1)	COHO	substrate diameter	area of overhang. rip. veg.	season		0.76	76	54.7	HR	O,R	S.E. Alaska
Ptolemy *et al.* 1991	162.1	COHO	fish wgt.	alkalinity			0.68	64	41.5	SM	R	diverse eco-regions of B.C.
Barber *et al.* 1981	61.2	CT	channel width	bank stability			0.56	76	40.9	HI	R	S.E. Alaska
Barber *et al.* 1981	61.3	DV	pool width	riffle width			0.54	76	39.4	HI	R	S.E. Alaska
Jowett 1992	202.1	BNT	water temp.	mean/median flow	% lake area	%flat slope	0.44	89	37.6	RP	R	New Zealand
Oswood & Barber 1982	105.3	DV	surface area	area w/ forest debris in riffles			0.49	76	35.8	HR	O,R	S.E. Alaska
Oswood & Barber 1982	105.2 (61.4)	COHO	gradient	area w/ depth <0.5m, velocity <0.3m/s	area of overhangriparian veg.	area undercut banks	0.49	76	34.8	HI	O,R	S.E. Alaska
Lanka *et al.* 1987	83.1	BNT, RBT, BKT, CT	elevation	relief ratio	drainage density	avg. reach width	0.51	65	30.6	RP	O	Colorado & Missouri river drainage in Wyo.
Oswood & Barber 1982	105.4	trout	area w/ forest debris in riffles	area w/ depth >0.5m, velocity >0.3m/s	area w/ forest debris in pools	area of overhangriparian veg. in riffles	0.43	76	30.5	HI	O,R	S.E. Alaska

Ptolemy *et al.* 1991. (models 162.1, 162.2)

In the stream habitat category Ptolemy *et al.*'s relationship predicting salmonid density based on fish weight, stream alkalinity and suspended sediment concentrations (model 162.2) was the highest ranked based on the R^2*DF indicator. The model predicting coho density (162.1) also ranked very highly. An advantage of these models is that the independent variables are measured with reasonable accuracy compared to independent variables used in other stream models (Oliver 1994). Measurements of commonly used habitat variables such as % cover can vary considerably between survey crews which results in poor repeatability. *Because the general salmonid model (162.2) predicts total biomass by using mean size per size category, its applicability can be extended to resident trout streams with multiple species complexes (Oliver 1994). The model can be used to compare measured production estimates to gauge the status of particular index sites.*

Perhaps the key limitation of Ptolemy et al.'s approach is that it only predicts fish abundance in "prime" or suitable habitat. The data used to develop the models included very few adult trout (>20 cm) density estimates. Oliver (1994) has shown that a large number of stream transects are required to use the model in conjunction with WUA to estimate reach-wide or stream-wide salmonid standing crop capacity. *Oliver (1994) points out however, that this method could be useful for annual assessments of stock status within discrete replicated sample sites. There is concern within the Fisheries Branch regarding the screening criteria used to select data to develop the regressions and the statistical approach used to develop the models.* The authors used only 20% of the total records with the highest densities. This arbitrary criteria was used to infer that these observations represented maximum salmonid densities (*e.g.*, carrying capacity). No other support for this data screening procedure was provided, and it is anticipated that the analysis will be repeated using more defensible criteria within 6 months (M. Labelle, pers. comm.). While these issues raise concern over the models as currently parameterized, the revised models will be potentially valuable. *The advantage of using an "in-house" method is that the assumptions and limitations are clearly understood within the Fisheries Branch relative other models where only published information is available. Because the input requirements of these models are compatible with the existing biophysical inventory of B.C., these models are widely applicable throughout the province at the reconnaissance level.*

Fraley and Graham 1981 (model 69)

This model predicts cutthroat and bull trout standing crop in Montana tributaries of the Flathead River drainage. This model uses 4 independent variables: overhead cover; instream cover; stream order; and, substrate. These variables are currently measured by the Province. About 2/3 of the stream reaches in the study were located in protected areas and were therefore not affected by development. The remaining 1/3 has been impacted to some degree by road building or logging. The investigators compared predicted and observed standing crop in 23 tributary reaches not included in the dataset (but within the Flathead system) used to develop the model. Predicted and observed densities at these sites were well correlated. Estimates of overhead and instream cover measures used in this model will vary between individuals, which may limit its predictive ability. *This model falls into the reconnaissance or intensive management category and could potentially be used (following validation in B.C.) for assessment of logging impacts for inland rivers containing cutthroat and bull trout.*

Sekulich 1980 (model 114)

As described by Fausch *et al.* (1988), this model predicts the standing crop of chinook age 0 in pools in several Idaho streams using 3 independent variables. The 3 variables used in the model were not specified by Fausch *et al.*, but are reported to be some combination of: number of eggs deposited, channel morphometry, flow, temperature, and biological variables. The number of eggs deposited was reported as the most important variable in all the models Sekulich developed. *Although the egg deposition input requirements of this model would limit its potential application, and clearly places this model in the intensive management category, two sites could be compared under an assumed egg deposition. This would potentially provide a comparative assessment between sites*. Readers wishing to apply this method should obtain this publication to assess the model more thoroughly, as it was not available to us during our assessment.

Oswood and Barber 1982 (models 105.1-105.4)

Oswood and Barber (1982) describe a stream survey technique and model which predicted fish abundance in 1st to 3rd order streams in the Tongass National Forest in southeastern Alaska. Individual models were developed for coho age 0, coho age 1+, Dolly Varden, and trout using 3 to 4 variables. Variables used in the models were: available spawning area (gravel size); area of overhanging vegetation; season (days since June 1); gradient; area of shallow slow water; area of deep fast water; area of undercut banks; and stream size. Their technique was based on diagrammatic maps of stream sections (of streams 1-30 m in width) emphasizing measurement of stream features rather than subjective judgements so that consistent results were produced over time and in relation to other survey crews. This technique was recommended by the investigators as especially appropriate for remote areas because: it provides a visual record of stream habitat; minimal time is required for the survey; and, time consuming and expensive laboratory analysis or time-series data are not required. *These models falls into the reconnaissance management level (and potentially the overview level) since the input requirements are minimal but on-site data collection is needed. The approach is particularly appropriate for remote areas and in situations where a visual record of stream habitat is required for management or legal purposes. These data are currently collected by the Province but it is unknown whether similar mapping techniques are employed.*

Barber et. al. 1981 (models 61.2, 61.3)

As reported by Fausch *et al.* (1988), Barber *et al.* (1981) developed regression models for Dolly Varden and cutthroat trout in southeastern Alaska using the transect methods described for Oswood and Barber 1982. The variables used to predict cutthroat trout (model 61.2) were channel width at bankfull flow and bank stability, and those used to predict Dolly Varden standing crop (model 61.3) were pool width and riffle width. *As discussed above, these models fall into the reconnaissance management level (and potentially the overview level) since the input requirements are low but do require on-site data collection. These data are currently collected by the Province but it is unknown whether similar mapping techniques are employed.* Readers wishing to apply this method should obtain this publication to assess the model more thoroughly, as it was not available to us during our assessment.

Jowett 1993 (model 202.1)

Jowett presents 4 models predicting brown trout standing crop in New Zealand streams from various combinations of hydrological, catchment, water quality, biological, and physical variables. The top-ranked Jowett model (202.3) uses water temperature, % WUA, instream trout cover grade, and gradient. Although many other authors have shown that IFIM methodology cannot be used to predict standing crop, Jowett validates his predictions. Given the many examples which demonstrate the problems inherent in this approach (discussed in detail in Section 3), Jowett's model (202.1) which does not include WUA, but which ranks closely behind the WUA model would seem to be a safer approach. This relationship uses water temperature, ratio of mean/median flow, % lake area, and % flat slope. *This model does not require expensive instream surveys, but does require some on-sight data collection, and therefore falls into the reconnaissance category. Because of the few brown trout streams in B.C. (Adam and Cowichan rivers), the model would have limited application in the province.*

Lanka *et al.* 1987 (model 83.1)

Relationships between fish standing stock and stream habitat and geomorphological measures were examined in high elevation (coniferous) Rocky Mountain streams in Wyoming. Estimates of standing stock included rainbow, brown, brook, and cutthroat trout longer than 100 mm. The dependent variables for the forested watershed model with the highest predictive abilities (model 83.1) were reach elevation, relief ratio, drainage density, and average reach width.

The principal advantage of Lanka et al.'s approach is that it has the potential to estimate standing stock using map-based information available for B.C. (either from TRIM maps or the B.C. Stream Atlas). This makes it a useful capability model at the overview management level despite its relatively low R^2 (0.51). The map-based information used by Lanka *et al.* was derived from 1:24,000 or 1:62,500 scale maps. The authors provide sufficient detail on how to derive their map-based and stream habitat measures. Although the models were developed using data from Rocky Mountain streams there is no indication of how the fish sampling sites were selected, so we do not how representative the relationships are across different habitats.

2.4.1.2 Model Ranking By Use Category

The habitat capability models were also ranked within each of the four model *use categories* (stock management, recreational/regional planning, habitat impact and mitigation, and habitat restoration and improvement). The 5 top-ranked models in each category are summarized in Tables 2.3 - 2.6. For brevity, we have provided detailed descriptions for only the top 2 models/papers from each category.

Table 2.3. Summary of the 5 top-ranked stream capability models within the **Stock Management category** to compare measured fish population status with predicted capability. Management levels: O=Overview, R=Reconnaissance, I=Intensive. See Section 2.3.2 for species abbreviations.

Author	Mod. #	Spp.	Independent Variables				R^2	N	R^2DF	Man. Lev.	Area Developed
			1	2	3	4					
Ptolemy *et al.* 1991	162.2	salmonids	fish wgt.	alkalinity	fixed non-filter. residue		0.84	1592	1327.6	R	diverse eco-regions of B.C.
Ptolemy *et al.* 1991	162.1	COHO	fish wgt.	alkalinity			0.68	64	41.5	R	diverse eco-regions of B.C.
Rosenau & Slaney 1983	54.1	mixed	area of cover/tot. area	nitrate			0.92	24	20.2	R	data from mainly Wyoming w/ a few B.C. streams
Jowett 1993	202.2	BNT	water temp.	benthic invert. bio-mass			0.45	42	17.9	O,R	New Zealand
Milner *et al.* 1985	49.7	AS, BNT	water hardness	mean width	mean depth	depth variance	0.53	38	16.9	I	Wales (5th variable is %46-60cm depth)

Table 2.4. Summary of the 5 top-ranked stream capability models within the **Recreational/Regional Planning category**, where coarse estimates of stock capacity are estimated from map-based or easily obtained information. Management levels: O=Overview, R=Reconnaissance,

I=Intensive. See Section 2.3.2 for species abbreviations.

Author	Mod. #	Spp.	Independent Variables				R^2	N	R^2DF	Man. Lev.	Area Developed
			1	2	3	4					
Jowett 1992	202.1	BNT	water temp.	mean/ median flow	% lake area	%flat slope	0.44	89	37.6	O,R	New Zealand
Lanka *et al.* 1987	83.1	BNT, RBT, BKT, CT	elevat ion	relief ratio	drainag e density	avg. reach width	0.51	65	30.6	O,R	Colorado and Missouri river drainages in Wyoming
Marshall & Britton 1990	24.2	COHO	stream length				0.88	24	19.4	R	Pacific Northwest coastal streams, ponds and side channels. Mainly sites on Vancouver Island
Lanka 1985	82.1	T	basin perime ter	reach gradie nt	mean basin elev.	width :dept h ratio	0.64	26	13.4	O	rangeland streams
Chisholm & Hubert 1986	65	BKT	mean depth	mean width	section gradien t	width :dept h ratio	0.69	24	13.1	R	Snowy Range of Wyoming

Table 2.5. Summary of the 5 top-ranked stream capability models within the **Habitat Impacts and Mitigation category**, where predictions of impacts of logging and flow reductions are required. Management levels: O=Overview, R=Reconnaissance, I=Intensive. See Section 2.3.2 for species abbreviations.

Author	Mod. #	Spp.	Independent Variables				R^2	N	R^2DF	Man. Lev.	Area Developed
			1	2	3	4					
Sekulich 1980	114	CHIN	3 variables (not specified) are some combination of number of eggs deposited + channel morphometry, flow, temperature, and/or biological variables				0.89	80	67.6	I	Idaho
Barber *et al.* 1981a	61.2	CT	channel width at bank-full	bank stabilit y			0.56	76	40.9	I	Southeastern Alaska

			flow								
Barber *et al.* 1981a	61.3	DV	pool width	riffle width			0.54	76	39.4	I	Southeastern Alaska
Barber *et al.* 1981a	61.4	COHO	gradient	area w/ D depth <0.5m, velocity <0.3m/s	area of overhang. riparian veg.	area of undercut banks	0.49	76	34.8	I	Southeastern Alaska
Oswood & Barber 1982	105.4	trout	area w/ forest debris in riffles	area w/ depth >0.5m, velocity >0.3m/s	area w/ forest debris in pools	area of overhang. rip. veg. in riffles	0.43	76	30.5	I	Southeastern Alaska

Table 2.6. Summary of the 5 top-ranked stream capability models within the **Habitat Restoration and Improvement category**, where predictions of benefits from managed changes in habitat complexity (restoring large woody debris in historically logged or channelized streams, adding boulder clusters, and stream fertilization) are required. Management levels: O=Overview, R=Reconnaissance, I=Intensive. See Section 2.3.2 for species abbreviations.

Author	**Mod. #**	**Spp.**	**Independent Variables**				**R^2**	**N**	**R^2DF**	**Man. Lev.**	**Area Developed**
			1	**2**	**3**	**4**					
Fraley & Graham 1981	69	CT, BULL	overhead cover	instream cover	stream order	substrate	0.64	134	83.2	I	Flathead River drainage
Barber *et al.* 1981 a	61.1 (105.1)	COHO	substrate diameter	area of overhang. rip. veg.	season		0.76	76	54.7	I	Southeastern Alaska
Oswood & Barber 1982	105.3	DV	surface area	area w/ for. debris in riffles			0.49	76	35.8	I	Southeastern Alaska
Binns & Eiserman 1979	36	T	5 variables (not specified) describing drainage basin, channel morphometry & flow, habitat structure (bio./phys./chem.)				0.97	29.1	38	I	Wyoming
Nickelson *et al.* 1979	101.1	CT	1 variable, "total cover," calculated from the abundance of suitable depths, instream cover, overhanging cover, surface turbidity, and velocity refuge				0.91	24.6	29	I	Oregon

Stock Management Category

Ptolemy *et al.* 1991. (models 162.1, 162.2) See description in Section 2.4.1.1 above.

Rosenau and Slaney 1983 (model 54.1)

A model predicting sustained standing crop (standing crop remaining after moderate fishing pressure) was developed for trout and char in streams. The regression was based on published data collected primarily from Wyoming and Wisconsin (n=39). Fish cover, nitrate concentration, total dissolved solids, alkalinity, and wetted width were found to be of predictive value. Using forward stepwise regression, the fish cover and nitrate explained 92% of the variability in standing crop (n=24). *This capability model is only applicable in late summer - early autumn, the period when the data were collected. As most streams in B.C. are phosphorous limited, this model should only be applied to streams that are nitrogen limited, which are typically lake headed. The model could be applied at the reconnaissance level where water chemistry is sampled and cover roughly estimated (Aquatic Resources Ltd. 1993).*

Recreational/Regional Planning Category

Jowett 1993 (model 202.1) See description in Section 2.4.1.1 above.

Lanka *et al.* 1987 (model 83.1) See description in Section 2.4.1.1 above.

Habitat Impact and Mitigation Category

Sekulich 1980 (model 114) See description in Section 2.4.1.1 above.

Barber et. al. 1981 (models 61.2, 61.3) See description in Section 2.4.1.1 above.

Habitat Restoration and Improvement Category

Fraley and Graham 1981 (model 69) See description in Section 2.4.1.1 above.

Barber et al. 1981a (model 61.1) See description in Section 2.4.1.1 above.

2.4.1.3 Additional Models

Following the technical workshop, a few capability models were identified by participants as having potential applicability in B.C. These models were either not included in our review prior to the workshop, or did not fall within the top-ranked group according to the R^2*DF indicator. Below we briefly describe these models.

Keogh Steelhead Model (P. Slaney, pers. comm.)

This model, when revised, could be used to predict mean annual steelhead parr and smolt yields, and perhaps juvenile production from nursery recruitment tributaries to lakes and rivers in the interior. This model falls within all the model use categories, but may be especially applicable to the habitat restoration and improvement category. The model's independent variables include, boulders, over-stream cover, and in-stream debris. Correlations (R^2) between age-specific steelhead densities and these variables ranged from 0.60 to 0.93. The data are currently being re-analyzed, using smaller subsets of independent variables to fill out the regressions for habitat units other than riffles. Sample size for the initial model was 116 habitat units sampled over 4 years. The number of streams with smolts and TDS, alkalinity, or nutrients has increased considerably since 1981, so the revised model should be more robust. A negative relationship between parr density and stream width will be used to permit expansion to larger streams since parr do not use mid-channel area in large streams. Some testing of the revised model is anticipated.

Binns and Eiserman, 1979 (Model 36)

This model predicts trout standing crop based on late summer stream flow, annual stream flow variation, maximum summer stream temperature, a food index, a shelter index, nitrate concentration, cover, eroding stream banks, substrate, water velocity, and stream depth. This model ranked poorly according to our R^2*DF indicator (29.1), principally due to the small sample size used in the analysis (n=38), especially in relation to the large number (8) of independent variables. *One of the major criticisms of the model is that it consists of subjective ratings which are converted into logarithmic values*. However, this model was applied to the Nechako River and did well. It predicted 14.7 kg/ha trout compared to the 12 kg/ha wild trout measured (after fishery closure) and the 3 kg/ha hatchery trout planted (1 yr after stocking yearlings) or 15 kg/ha in total. *This model is applicable to nitrogen-limited streams only, especially the interior inland rivers and lake outlet streams (and not far from Wyoming) and could be used as a secondary method for stock management, recreational/regional planning, and habitat impact assessment until an improved methodology is developed in B.C.* See the Habitat Quality Procedures Manual by Binns (1982) for a detailed description of the model and how to use it.

Marshall and Britton, 1990 (Model 24)

Simple linear and curvilinear regressions were used to relate coho smolt yield and rearing space (stream area or length) based on data collected from 21 streams, 2 ponds, and 2 side channels within B.C. The objective was to determine carrying capacities based on stream size. The authors concluded that:

- the curvilinear equation was a better descriptor of the yield - rearing space relationship than the linear form when coho numbers or biomass was examined in relation to stream area;
- smolt biomass was a better measure of carrying capacity that total number; and
- for small streams (<4 km in length or <20,000 m^2 in area), area was more representative of carrying capacity than length. Overall however, stream length proved to be a better predictor of capacity than area.

The model could be useful at the recreational/regional planning level for coho *habitat capability. No indices of stream productivity were incorporated in the original model, however inclusion of these parameters could potentially increase the predictive power of these regressions.* Dr. Mike Bradford (Department of Fisheries and Oceans, West Vancouver) has extended the approach of Marshall and Britton by increasing the size of the data set and including additional explanatory variables. His analysis included 99 streams from B.C., Washington, Oregon, and California. In addition to stream length and area used in the Marshall and Britton analysis, latitude, water yield (mean annual discharge/watershed area), a minimum flow index, and valley floor slope were examined for their ability to predict coho smolt production. These additional variables did not increase the predictive ability of Marshall and Britton's original models. The best model using the extended data set predicted coho smolt production based solely on stream length (R^2=0.7). The analysis is currently being reviewed within DFO and will be prepared for publication in the fall.

B.C. M.E.L.P Steelhead Production Modelling

The Fisheries Improvement Unit has been involved in the development of steelhead production models for application to B.C. streams. The objective of these models is to estimate the production capacity and eventually harvestable surplus of all steelhead streams in the province for management purposes. The following information was taken from an unpublished report prepared by Ron Ptolemy ("Present status of steelhead production modelling," Fisheries Improvement Unit, June 1987)

The overview approach used drainage basin area or mean annual discharge (M.A.D.) estimated from maps to predict smolt yield capacity. 28 streams were used in the development of the models. *The advantage of this approach is that it gives a production estimate to use as a reference which is easily obtainable at low cost. The disadvantages are that fish distribution, habitat quality, or productivity differences between streams are not addressed.*

The more detailed approach uses a step-wise methodology which addresses the distribution and abundance of juvenile rearing habitat and populations based on stream size and flow. The method is step-wise in the sense that it uses various relationships to link topographic mapping to habitat and fish populations. The 4 steps in the model are:

1) determine likely distribution of steelhead rearing habitat based on stream order, M.A.D., or water yield (information obtained from maps);
2) estimate wetted area of rearing habitat using M.A.D. and stream width relationships based on stream length, M.A.D., or water yield;
3) estimate fry and/or parr populations using either average fish density and stream width relationships, or weighted useable area predicted from summer flow stage and stream alkalinity;
4) estimate smolt yield by applying survival rates from smolt age and survival relationships based on stream temperature or smolt age.

Tautz *et al.* (1992-Appendix B, model 1.7) applied this approach to estimate steelhead carrying capacity for the Skeena River.

2.4.2 Lake Models

Our ranking of lake models distinguished two methods which are both applicable at the overview management level (Table 2.7). Because the fisheries branch has two in-house process models (Large Lakes Kokanee Model, Small Lakes Integrated Management Model) which will be used for evaluating policy options at the intensive management level, we did not attempt to evaluate empirically-based intensive management models which ranked poorly (due to low sample sizes) in our procedure (Appendix A). All lake capability models fell within the Recreational/Regional Planning model use category.

Table 2.7. Summary of top-ranked lake capability models Management levels: O=Overview, R=Reconnaissance, I=Intensive. See Section 2.3.2 for species abbreviations

Author	Mod. #	Spp.	Independent Variables				R^2	N	R^2DF	Man. Lev.	Area Developed
			1	2	3	4					
Godbout & Peters 1980	1.5	BKT	TP	effort	area		0.88	66	58.0	O,R	Laurentian Shield lakes (Quebec)
Scarborough & Peters (unpubl.)	201.1	mixed	effort	depth	TP		0.85	46	35.7	O	lakes of various trophic levels lying on both igneous and

											sedimentary drainage in Ontario

Godbout and Peters 1988 (model 1.5)

The best model selected from Godbout and Peters (1988) predicts stable catch of brook trout in Laurentian Shield lakes in Quebec from total phosphorus (TP), fishing effort, and lake area. A number of other models from this paper were ranked higher in our scoring system, but they either used regressor variables whose ranges are not applicable to B.C. (*e.g.*, altitude), or additional variables which added little explanatory power to the models but reduced their application potential (*e.g.*, required mean weight). *The principal disadvantage of using Godbout and Peters model(s) in B.C. is that rainbow trout, rather than brook trout, are the dominant sportfish in the province.*

Godbout and Peters' (1988) models were validated using an Ontario mixed species dataset (Scarborough and Peters, unpubl.). In this validation, the model predictions were well correlated with observed catch, but were consistently higher. Unlike most other lake capability models, Godbout and Peters' data set consisted entirely of small lakes (<100 ha) without commercial fisheries and which had not been stocked. The authors only used data from lakes from which at least 5 consecutive years of catch statistics had been collected, and where no significant trends in catch were detected. *This data screening procedure minimizes parameter bias inherent in other lake capability models and provides predictions which are more indicative of long-term stable catch. This model would seem useful for determining where to manage among the 1000's of small lakes in B.C. and is applicable at the overview (or reconnaissance) level. The model should not be used to predict absolute stable catch estimates as shown by the Scarborough and Peters (unpubl.) analysis, but may be used to compare relative fisheries potential among different systems.*

Scarborough and Peters, unpublished (model 201.1)

The model selected from Scarborough and Peters (unpubl.) predicts sportfish catch based on angler effort, TP, and mean depth. Two other models from this paper using different combinations of these regressor variables (*e.g.*, effort and TP, effort, TP-area, and depth) rated very closely to this model. Scarborough and Peters' (unpubl.) dataset consisted of 46 Ontario lakes whose fisheries were comprised mainly of smallmouth bass and lake trout with minor components of rainbow trout, whitefish, and northern pike. *This model may be more appropriate for predicting rough estimates* *of absolute catch in mixed species lakes (overview level) than Godbout and Peters (1988), especially in northern lakes larger than 100 hectares*. This paper is currently an unpublished manuscript, and we are unaware of its status vis-a-vis publication.

2.5 Recommendations

Below we provide some recommendations concerning the immediate use of the top-ranked models in B.C., as well as suggestions for the development of future models using data collected within the province.

1. Experience has shown that models applied outside of the geographical area where they were developed often give poor predictions. If the models are to be used prior to validation, users must ensure that the range of independent variables in the target system is within the range used to calculate the regression. At this stage, the models should be used only to achieve "ballpark" estimates which need to be confirmed from field observations or alternative assessment methods.

2. Use provincial databases to develop predictive relationships based on B.C. data. Information in the B.C. lakes database together with Small Lakes Integrated Management catch data could be used to develop lake overview capability models relatively easily. Development of stream capability models will require more effort in terms of data synthesis. The B.C. Stream Atlas and TRIM maps combined with fish abundance data could be used to generate predictive relationships at the overview level. Existing methods such as Ptolemy et al.'s (1991) model could possibly be enhanced by adding additional explanatory variables such as stream gradient or elevation.

3. Our system of ranking and evaluating the models is just one of many possible methods for selecting the best models specific to the scope of this project. An analyst with different needs and priorities should review the database tables in Appendix A (available as an EXCEL 4.0 file) using alternate criteria to identify other useful approaches. As Fausch *et al.* (1988) recommend, stratify the analysis of regression models by regions of homogeneous climate, geology, landform and soils to reduce residual variance.

4. The intensive capability models reviewed here are considered inadequate for the management of high-value resources because the models have not been validated in B.C. It is recommended that experimental manipulations be used to evaluate the impact of watershed disturbances in these situations.

5. Assess the predictive ability of the models using non-parametric methods. Estimates of model prediction variance are very sensitive to linear regression assumptions. Bootstrapping and jacknife approaches can provide confidence limits and other uncertainty measures which are not dependent on these assumptions. For example, the jacknife method involves estimating model coefficients after deleting each record sequentially and replacing the record with a another randomly selected point from the dataset. For each set of model coefficients, the residual for the predicted point which was deleted is calculated. This procedure is repeated for all records in the dataset. The distribution of residuals from this analysis provides a good measure of the prediction variance which reflects the underlying distribution of the data.

6. Validate the models by applying them in systems which were not included in the development datasets and compare predicted and observed values. For initial model testing, estimate model coefficients using a fraction of the total dataset (say 50%) and test the model predictions using the remaining fraction (cross-validation).

3.0 A BRIEF REVIEW OF PROCESS MODELS IN REFERENCE TO ESTIMATION OF HABITAT CAPABILITY

This chapter will provide an overview of the different types of process models which can be used to predict habitat capability in freshwater habitat. In the context of this review, we define process models as those models which simulate fish populations, bioenergetics and growth, or ecosystem-trophic processes. Like the empirical models which were ranked in the Section 2, process models use input data in some form to make a prediction, and in many cases process models can be made up of a set of linked empirical regressions. However process models can also be composed of relationships which have a underlying theoretical, rather than empirical basis. Process models can be deterministic or stochastic in nature.

The intent of this review is not to summarize all process models which can be used to estimate habitat capability in freshwater systems. Rather, our goal is to review a representative subset of available process models in order to demonstrate various approaches which have been used to model capability. The strengths and data requirements of these general approaches will be summarized but we make no attempt to rank the process models in terms of their applicability for estimating capability in B.C. streams and lakes. The use of process models will be productive only if those making management decisions engage in the modelling process directly (Walters 1986). That is, if they contribute to the conceptual structure and formulation of the model, as well as participate in gaming and analysis using the model. There are three basic reasons why process models which are to be used for management purposes must be "home-grown":

1. In order for model users to understand and trust the model, they must have some input into its development;
2. management policies, the spatial and temporal scales of information to be used as input to the model, and the required resolution of predictions are generally quite specific to individual management agencies; and
3. the underlying structure and functional relationships used in the model must reflect the hypotheses of the managers/scientists who are making policy decisions, or providing advice for these decisions.

There are many good reasons to build process models. The development and use of process models can be used to:

1) increase understanding of the system by making mental models explicit;
2) improve communication among researchers and managers;
3) identify knowledge gaps;
4) explore alternative hypotheses concerning system response;
5) help design management experiments;
6) assess/refine experimental designs;
7) provide teaching/training; and
8) assess the implications of alternative management policies or mitigation options.

This discussion will focus on the last item of this list, that is, the predictive capabilities of process models. It should be noted that in many cases the most beneficial product generated from a modelling exercise is not the predictive tool, but the insights and synthesis gained through the development of the tool and the model gaming/analysis which often follows.

The value of process modelling in fisheries management is to provide clear caricatures of nature against which to test and expand experience. This is the key advantage of process models over simple regression models. Take for example, the situation where a fisheries biologist is trying to evaluate the potential of a lake to support a put-and-take fishery based on a prediction of carrying capacity. The biologist retrieves basic physical-chemical information from a centralized database and plugs the lake's MEI into a regression which predicts biomass. Based on a promising result, the biologist surveys the lake to verify his prediction. To his surprise, biomass is much lower than predicted. At this point all that the biologist can do is come up with a list of hypotheses explaining why biomass in this system is so much lower than the predicted value based on MEI. Using the empirical regression, the biologist has no way of evaluating the relative likelihood of the different hypotheses explaining the low standing crop in the lake. However, if the biologist used a process model, he/she would have a tool to evaluate the different hypotheses. For example, he/she could ask, "how low do survival rates have to be to reproduce the observed abundance estimate?" These types of analyses help the biologist narrow down the range of likely hypotheses to help explain what is happening in the system. This process of simulated hypothesis testing encourages the biologist to develop management experiments to help test these hypotheses. In addition, the biologist can explore different stocking rates using the simulation model and compare his projections with those based on standard provincial stocking formula. Hopefully, this scenario has demonstrated how much more useful, from a decision making perspective, a process model can be relative to a empirical capability regression model.

The process models we reviewed were classified into 3 groups according to the objectives for their development:

1) to evaluate fisheries policy;
2) to estimate the impact of watershed disturbances; or to
3) improve understanding of ecological processes.

Below we discuss the general advantages and data requirements of a selection models in each of these 3 categories. In total, 27 models were reviewed, and abstracts or summaries of these models are provided in Appendix B. 11 models were selected from the grey literature and 16 were selected from an electronic search on the BIOSIS Database (1991-Sept.1993). The following search parameters were used (* denotes wildcard):

(TROUT ***or*** SALM* ***or*** FISH*) ***and***
(STREAM* ***or*** RIVER* ***or*** LAKE*) ***and***
(PROCESS ***or*** ECOSYSTEM ***or*** STOCHASTIC **or** (LIFE HISTORY) ***or*** PRODUCTION) ***and*** (MODEL* ***or*** SIMULATION)

The numbering sequence of citations used in this chapter corresponds to the numbers beside each reference in Appendix B.

3.1. Models for Evaluating Fisheries Regulations

Of the 3 categories, models within this group are most homogenous in terms of their structure and objectives. Typically, these are age-structured models with seasonal or annual timesteps where survival, recruitment, and sometimes growth are simulated. These models provide more than an estimate of habitat capability; they often simulate age-structured abundance and growth responses of populations to management actions and the subsequent effects on angling quality. As such they are a much more useful interpretive tool compared to individual empirical models since it is often the overall population response and its effect on angling, and not just the capability of a given system, which determines the success or failure of specific policy options. The objectives of the management-based process we reviewed are listed below:

1.1	Evaluate the viability of Colorado trout populations across a range of size and slot limit regulations and angling intensities.
1.2	Assess alternative harvesting and stocking strategies on catch rates, growth, and effort of rainbow trout at both single-lake and regional (multi-lake) scales
1.3	Examine kokanee growth and density responses to changes in lake trophic status, predator and competitor abundances, and alternative stocking rates and harvesting regulations of kokanee and predators.
1.4	Assess the potential effects of a number of management actions on single fish populations.
1.5	Assist fisheries biologists and managers in making decisions regarding the management and assessment of lake trout populations in Ontario.
1.6	Assess salmon population viability in the Columbia River System in relational to natural escapement and hatchery broodstock needs
1.7	Estimate steelhead carrying capacity for the Skeena River
1.8	Predict the response of native steelhead to long-term supplementation with hatchery fry and smolts.
1.9	Evaluate the potential for exploitation of northern squawfish to reduce predation rates on salmonid smolts.
1.10	Evaluate walleye stocking strategies in relation to release timing and size at release
1.11	Evaluate the dependence of a perch population and yield on management and fishing rules.
1.12	Assess the effects of stocking and harvest regulations on walleye populations and fishery characteristics.

The data requirements for these management-based process models are dependent on both the stage of model development (*e.g.*, formulation, validation/calibration) and the spatial coverage over which the model will be applied. The initial development of relationships and overall model structure is usually based on experience with a few intensively studied systems. Data from these systems are used to define the shapes and baseline parameter values for key functional relationships in the model such as density-dependent survival and/or growth rates. For example, the prediction of growth rates in the Large Lakes Kokanee Model is based on a functional relationship which uses secchi depth and kokanee density-at-age. This function was parameterized using data principally from Idaho lakes which had been synthesized into a well documented and publicly available database. A similar synthesis of information specific to B.C. small lakes was undertaken to help develop/parameterize the key functions and rate constants of the Small Lakes Integrated Management Model. The construction of a database which synthesizes growth and survival information from intensively studied systems would greatly facilitate the development of future models, or more detailed analyses (*e.g.*, sensitivity analysis) of existing models used within the fisheries branch. Development of such a database is no trivial task; great care must be taken to ensure that data are corrected to account for differences in sampling methodology or timing of collection between systems. A synthesis of information on fishing quality (fish size and CPUE) and effort would greatly facilitate the analysis and modelling of angler response in response to regulations and population characteristics.

The second stage of model development usually consists of testing the model predictions and refining the structure. Given sufficient information, validation sites should be excluded during the development of the process model's functional relationships. The more detailed the validation dataset, the more rigorous the tests of model predictions. Ideally, these data reflect not only current conditions, but the response of the system to actions which are being modelled.

The third stage of model development consists of applying the model to target systems to evaluate specific policy options. A certain amount of data is needed to calibrate the model so that it reflects current conditions in the target system. Process models which are intended to be applied to many systems (*e.g.*, SLIM) should be designed so that the calibration data requirements are realistic relative to what data are currently available, or what will likely be available in the future. One of the advantages of process models is that they can be applied to systems with little baseline data if the user is willing to live with the uncertainty in the predictions.

3.2. Models for Impact Assessment and Evaluation of Mitigation Opportunities

Simulation models in this category are used to assess the impact of environmental perturbations on fish populations. The models reviewed reflect a wide range of perturbations, ranging from regional-scale impacts (global warming, acid rain) to very localized problems (*e.g.*, declining nutrient inputs to Kootenay Lake). These models do not always estimate habitat capability, and sometimes provide only an index of the impact's magnitude on a particular life stage or biological rate (*e.g.*, egg-fry survival as a function of area logged). It is often easier to accurately model the impact of a perturbation on an indicator such as juvenile survival rather than on habitat capability, since the latter indicator requires incorporating some method for simulating compensatory population responses. However, from a management perspective, models which predict the net effect of an impact on stock productivity or habitat capability are more useful, since they incorporate hypotheses concerning the extent of compensatory responses to the impact.

Another useful benefit of impact assessment models is their ability to assess alternate mitigation options. For instance, application of the Kootenay Lake Fertilization Response Model suggests lake fertilization will not improve kokanee abundance or growth because of the competitive effects of *Mysis*. This hypothesis has been helpful in managing expectations for the outcome of the experimental fertilization program, and has supported the development of a rigorous monitoring program. In some instances, impact-based models can help managers adjust fisheries regulations in relation to the effects of a perturbation on stock productivity or carrying capacity. The Atlantic Salmon Regional Acidification Model has been used to identify optimal stocking rates and harvesting regimes across systems of different acidity in order to meet target escapement goals. The objectives of the impact-based process models we reviewed are listed below.

2.1	Simulate long term changes in plankton, kokanee, and Gerrard trout populations in response to changes in lake fertility associated with phosphorous loading.
2.2	Partition the variability in adult returns between the effects of climatic variability in the stream and ocean, changes in stream conditions caused by logging, and variation in fishing mortality.
2.3	Simulate the effects of streamflow and sediment transport on survival of salmonid embryos incubating in spawning gravels.
2.4	Estimate monetary loss and fish loss of catchable trout due to livestock impacts.
2.5	Assess the recovery potential and mitigation opportunities for Atlantic salmon

	stocks depleted due to acid rain.
2.6	Assess the effects of electric power stations and hydropower plants on recruitment of walleye.
2.7	Simulate the impact of global warming on the production of spring chinook salmon in the Columbia River System.
2.8	Evaluate the effectiveness of management actions that are designed to improve migration survival of smolts in the Columbia River System.
2.9	Assess the potential effects of temporal changes in carrying capacities on a fishery resource resulting from a proposed hydroelectric project.

The data requirements of impact-based models are more variable relative to the management-based ones, principally due to the variety of modelling approaches employed. The temporal and spatial scales of the models are often finer since they must capture the appropriate scale for the dynamics of the impact being simulated. Because of this, the number of sampling locations and frequency of samples required as input or validation will often be higher. In addition, good contrast in validation/calibration datasets are required to test the model's ability to simulate the extent of an impact and the potential for recovery. This includes estimates of the change in predicted and model input variables upstream/downstream or before/after the perturbation in question. For example, the time course change in TP loadings in Kootenay Lake provided a good dataset to test the Kootenay Lake Fertilization Response Model which predicts the trophic response to changes in nutrient loading.

3.3. Models to Increase Mechanistic Understanding of Ecological Processes

The variety of models developed to gain insights into ecological mechanisms is limitless. We reviewed a tiny sample of models in this category to provide some examples of the breadth of questions being addressed. The objectives of the ecologically-based process models that we reviewed are listed below.

3.1	Evaluate the capture rate and timing of prey captures on juvenile salmonids by norther squawfish.

3.2	Assess the relative influence of deterministic *vs.* stochastic process on stream community structure.
3.3	Quantify the effects of diet shifts on growth rates of chinook salmon.
3.4	Assess the effect of growth rate on recruitment variability in walleye populations
3.5	Evaluate the efficiency of compass movement as a migratory mechanism in the coastal homeward migration of Fraser River sockeye salmon.
3.6	Quantify the effect of forage fish community structure on piscivore production based on a biomass size spectrum approach.

Although the insights gained by such modelling exercises may ultimately lead to improved management, models within this category cannot generally be used to directly assess management policy, the impact of watershed disturbances, or habitat capability. The data needs for the development of ecologically-based process models are as varied as the objectives of the models, and we feel they are outside the scope of RIC data inventory concerns.

3.4 Recommendations for Future Modelling Efforts

This review has highlighted the benefits and data requirements of process models for estimating habitat capability and evaluating fishery management options. This review can also be informative to those considering the development of future process models. Below we provide some recommendations concerning process model development and analysis. These recommendations are based on our direct experience with modelling projects within the B.C. Fisheries Branch as well as within other agencies (*e.g.*, Department of Fisheries and Oceans).

Consider the Detail of Model Structure Carefully

One of the greatest challenges in systems analysis applied to resource management is deciding on the required level of complexity for the model. The spatial and temporal resolution of the model and its underlying structure should be determined based on the questions being asked. The tendency is to put in everything we know, even if some of the details are irrelevant to the problem being considered. Be aware that by increasing model complexity unnecessarily, you may:

1) reduce the portability of the model to other lakes/rivers;
2) reduce the number of potential model users;
3) hamper the development of sensitivity analyses which incorporate the effects of model uncertainty into policy projections; and
4) increase the chance of not detecting errors in model formulation or programming.

The challenge for those designing the model is to develop a tool that is sufficiently sophisticated to deal with most of the variation in biological systems and policy options, while at the same time being simple enough for routine application by decision-makers.

Choose Your Target Audience Carefully

Developing a model to be used by all of the people, all of the time comes at a considerable cost. Generally, the greater the number of potential users, the more difficult it will be to keep the model simple. Understandably, everybody wants their pet process or policy action included. On the other hand, if a simulation model is to be used by a wide-range of biologists and decision-makers, their input is critical if you want them to trust and use the model. In this situation, be prepared to spend a considerable amount of time addressing the users' specific needs and pet ideas, some of which could provide valuable additions to the model.

In some instances, models are designed as tools specifically for research scientists to explore policy options for important assessment problems. Since this represents a much smaller group who are generally more comfortable with models, these tools tend to be more streamlined and require less resources in terms of 1) user-interface development and 2) including all possible policy levers for every situation which can be encountered within the province.

Don't Underestimate the Time for Model Debugging and Refinement

Model development is an iterative process. Only after the working model is produced can the designers see how well their representation of system processes mimic reality. After the reality check, refinements are usually needed to improve the model and increase its ability to simulate our observations. This process not only improves the structure of the model, but roots-out programming errors. The process of refinement and debugging requires close cooperation between the modeller/programmer and the core group of managers/scientists who are developing the simulation.

Don't Neglect the Importance of Incorporating Uncertainty into Policy Analysis

In a world of limited resources, modelling projects are often considered complete once the core group of model developers are happy with the software product. However detailed policy analysis which incorporates model uncertainty is critical for testing different management decisions. Rather than asking, "what is the single best stocking rate for Lake X based on the results of the most likely (*i.e.* baseline) simulation?" it may be more appropriate to ask, "which stocking rate (or set of stocking rates) is best across different assumptions about the true state of nature?" Different states of nature can include alternate representations of important processes (*e.g.*, strong density dependent growth *vs*. weak density dependence), or alternate scenarios which can explain the current condition of the system (high natural mortality *vs*. high fishing mortality *vs*. poor natural recruitment). From this type of analysis, one may decide that the best stocking policy is the one which is most robust to different possible states of nature, rather than the one which produces the best response based on the biologists guess of the most likely single state of the system.

Training Model Users Takes Time

The people who initiate modelling projects, and who spearhead the development of the model within an agency generally have some experience with the use of models in biology. However many of the potential users will not have this advantage and will require a certain amount of training before they feel comfortable including the model as part of their everyday toolbox for dealing with assessment problems. This should come as no surprise since the use of computers in resource management is a relatively recent occurrence. The core group of model builders should be prepared to invest the time and effort into training. The use of workshops has been the traditional way for training model users, however one-on-one strategies may be more effective. Members of the core modelling group should plan on travelling to regional offices where they can help set-up the model for local lakes/rivers and demonstrate how to game with the model. More detailed sensitivity analyses performed by the core group on specific regional problems will demonstrate alternative ways of using the model in policy analysis.

The training investment needs to be seriously considered by those thinking about incorporating a decision support system into the assessment process. Will the regional biologists have sufficient time to adequately understand and use the model? Are there adequate resources to support more detailed policy analysis? To get the most out of process models, those supporting the project need to understand that these issues are as important as the production of the software itself.

4.0 REVIEW AND SELECTION OF SUITABILITY MODELS

4.1 Overview and Analytical Approach

There are relatively few suitability models compared to the large number of potential capability models. The literature is dominated by discussion of the Habitat Evaluation Procedure (HEP) and the Instream Flow Incremental Methodology (IFIM), both of which were developed by the Western Energy and Land Use Team of the U.S. Fish and Wildlife Service based in Fort Collins, Colorado. HEP (Terrell *et al.* 1982) is considered a generalized method for examining habitat suitability in both lakes and streams using comparisons of "habitat units" which are indices of available habitat. It does not provide absolute measures of usable habitat. It is useful for relative comparisons of undisturbed conditions between ecosystems and for examining relative change in indices of available habitat at different times. It fundamentally relies on habitat suitability curves (those that describe the relative suitability of use by a given species and life stage over a range of values of a given parameter). These curves may or may not require new data collected from the field as they can be developed using existing information or "expert opinion". IFIM (Bovee 1982) is specifically used to explore effects of flow manipulations in streams. It is primarily a negotiating tool that can be used to examine the relative change in habitat indices associated with a flow manipulation and mitigation techniques. It is not used to define flow needs to protect a fish population. Like HEP, output is in terms of relative indices, not absolute units. Unlike HEP, IFIM can simulate changes in physical and chemical parameters that are affected by a change in flow. The method is fundamentally based on the use of suitability index (SI) curves for four variables (velocity, depth, substrate, and cover) for which the data must be collected in the field using a standardized protocol. Computer simulations are necessary to examine changes in physical habitat as a function of flow manipulation. Details on the respective uses of HEP and IFIM have been described by Armour *et al.* (1984). In summary they indicate that:

- HEP applies to all habitat types;
- IFIM only applies to streams in which flow manipulations are negotiable in a potential impact or mitigation;

- HEP can be applied where impacts are wide ranging, including changes in flow, chemical loading, temperature, etc.; and
- HEP and IFIM can be used in combination to develop mitigation recommendations that may include channel design.

HEP and IFIM have received unprecedented attention in the United States mainly due to the Water Policy Initiatives of President Carter (Wesche and Rechard 1980) in which maintenance of instream values was considered important. Implementation of those policies began in 1978 with the creation of 19 task forces, one of which was given the responsibility of addressing questions related to what are acceptable in-stream flows. A result of task force activities was that water flow was recognized as a fundamental component of planning for large project developments. The initiative produced an acute need for methods to explore effects of flow manipulation on fish habitat in freshwater systems. Although many methods were reviewed (Wesche and Rechard 1980), the main focus in the United States to the present is development of HEP and IFIM. IFIM in particular, is used almost exclusively for assisting with water management decisions, not only in the United States but now in Canada as part of large water flow manipulation studies and projects. Examples include:

- Several hundred applications indicated in Armour *et al.* (1984) and many more in NERC (1992);
- Interagency analyses of hydro developments in the Columbia Basin (K. Stein, US Forest Service, Portland, Or.);
- Analyses contributing to regulation of altered Columbia River flows (M. Henry, Fed. Energy Regulatory Commission, Portland, Or.);
- Kalamath Basin Adjudication Project (D. Ford, Portland, Or);
- Analyses by Northern California In-stream Flow Group (J. Steele, California Fish and Game, Environmental Services Branch);
- Kemano Completion Project (Triton Environmental Consultants, Richmond, B.C.); and
- B.C. Hydro Project Team for Fish Habitat Suitability (G. Matthews, Project Head, B.C. Hydro, Burnaby, B.C.).

In addition to the HEP and IFIM, only four other suitability models were found. The Missouri "Stream Habitat Evaluation Procedure (SHEP)" (Fajen and Wehnes 1981) combines an independent ranking of habitat quality parameters and habitat alteration parameters to determine a habitat quality index. Another approach has been called the

Fish Habitat Index (FHI) in which geomorphic parameters including streamflow, pool-riffle ratio, surface shading, pool quality, and riffle quality are used to determine a habitat condition score (Parsons *et al.* 1981). Habitat condition score (HCS) is weighted by reach length and summed to determine an overall HCS for a stream. No biological criteria are included. A habitat suitability index (HSI) for reservoirs that are being designed was proposed by McConnell *et al.* (1984). A composite score is developed from a series of primary "attributes" (temperature, turbidity, nonliving cover, drawdown, and shallow cove frequency). The value of each attribute is determined from a series of secondary attributes that are thought to determine the suitability of reservoir habitat for a given species. The secondary attributes are determined from "opinion", engineering plans, climate records, on-site inspections, and published literature. The composite score is a numerical index that is compared against a series of suitability tables that provide a measure of suitability to be used as a guide in the selection of design and operation strategies for proposed reservoirs. Finally, Tennant (1975) proposed a method for recommending flows necessary for the protection of fish habitat. Based on empirical relationships, the Tennant method recommends maintenance of flows above a critical proportion of mean annual flow to provide suitable habitat for fish.

Given these limited numbers of suitability models, a ranking that we applied to the capability models was not appropriate. Instead, we examined each model independently to examine how it works and determine if or how it may be applied to lakes and streams in British Columbia. Information was taken from the bibliography prepared by Aquatic Resources Ltd. (1993), personal communications with users of the various methods, and other references found in manual and computer assisted searches of the literature.

4.2 Methods Review

The first step in the selection of any suitability model is a scoping process in which study objectives are defined, the study area is delineated, and the fish species of interest is selected. A scoping process is specifically laid out for HEP and IFIM and although it is not identified in the other methods, it is a necessary step, mainly to avoid the wrong selection of a method.

For HEP, factors that potentially limit fish abundance in the given habitat should be included in the variable list that is compiled in the scoping process. Final indices that are generated by both HEP and IFIM result from the product of HSI or suitability indices (SI) with habitat area which assumes that habitat area has some relationship with fish abundance or production. Unfortunately, determination of whether area or other factors actually do limit fish abundance can be time consuming and may require site specific sampling or experimentation. Normally such effort is avoided, but in so doing, the results may be misleading. Bisson (1992) comments that identification of limiting factors is chronically avoided in impact or baseline analyses due to:

- excessive reliance on professional judgement in the absence of site-specific data;
- extrapolation in space and time using data from a remote site despite the fact that limiting factors are difficult to identify even in long term studies of salmonids (*e.g.*, Alsea watershed study, Carnation Creek study);
- focussing on only one part of life history when overall production may also be limited by other factors in another life stage;
- failure to consider important limiting factors in systems which are poorly understood; and
- oversimplification of relationships in complex systems.

All of these concerns are particularly important when an objective is to quantify fish abundance or production. However, suitability models are not intended to provide absolute estimates of fish abundance or unequivocal predictions. They are not intended to provide unequivocal functional relationships between limiting factors and indices of fish abundance. There would be no need to run a suitability model if that was the goal. Rather, suitability models must only be considered tools to explore scenarios of relative change in characteristics of fish habitat in relation to potential impact, mitigation, management action, etc.. In the scoping process, this limit to the capability of the model must be clearly established. For HEP, identification of factors that best enable the model to achieve the goal of relative comparison are what must be identified.

4.2.1 Habitat evaluation procedure (HEP

Detailed descriptions of HEP methods are provided by Terrell *et al.* (1982) and Armour *et al.* (1984). The following summary is taken from these reports.

After the initial scoping session, a habitat suitability index (HSI) is developed. HSI can either be empirical regressions, mechanistic models, or descriptions (a judgement call based on opinion, literature, or other data). Mechanistic approaches are most commonly used (Figure 4.1) and require the use of suitability index (SI) curves (Figure 4.2). A mechanistic model is structured as a tree diagram in which the variable at the end of every branch is thought or known to relate to the suitability of a given habitat for the given fish species and life stage. For example, in Figure 4.1, percent cover is represented by V_2 and percent pools is represented by V_1. V_1 and V_2 can contribute to the life requisites of cover and reproduction. For any one habitat variable there may be more than one symbol, each representing a different life stage (i.e., a separate symbol representing a separate SI curve for adult, juvenile, and fry for any one variable name). SI curves that are required for variable symbols are determined from literature sources, expert opinion, and field studies. The SI curves are then used to determine the value to assign to each variable symbol. For example, in Figure 4.2 (top figure), the SI for cutthroat trout in the 0-9cm size category would be about 53% if there was a mean depth of 30 cm.

The SI's are then aggregated to determine the HSI. This can be done in one of three ways. The average value method (AVM) simply calculates the geometric mean:

$$AVM = (V_1 \text{ x } V_2 \text{ x } V_{i...n})^{1/n}$$

The interactive limiting factor (ILF) method weights low SI's heavily which means all of the SI variables are considered equally important and if any receive a low SI value, that will pull overall suitability down. It is calculated as:

$$ILF = (V_1 \text{ x } V_2 \text{ x } V_{i...n})$$

Third is the lowest SI (LSI) method in which HSI is assigned the lowest SI score. This approach assumes that the variable having the lowest SI will limit overall habitat suitability.

The next step is to calculate habitat units (HU). The area of each habitat type that is used by the given species is measured and HU is then determined as:

$$HU = HSI \text{ x area}$$

HU's are simply compared in space for baseline estimates or site comparisons at any point in time, and through time as part of mitigation or impact assessments.

The application of HEP (and also IFIM (Section 4.2)) is based on an extrapolation of suitability indices to areal units. This calculation implies that a linear relationship exists between fish abundance and habitat area. However, Mathur *et al.* (1985), Scott and Shirvell (1987), Bisson (1992), and others, have shown this generally not to be true. Others have shown that it does exist on a site specific basis (Jowett 1992, others discussed in Section 4.2). Because of this disparity, it is clear that HSI may only be extrapolated to HA if the linear relationship has been established for the study area. Otherwise HSI should not be extrapolated to HA and HSI should be discussed as an index of potential suitability for the fish species and life stage of interest, not one of absolute suitability. The success of HEP is also dependant on accurate SI curves. Any error with these will be multiplied through to the determination of HSI. This concern is well known and certainly at the intensive level of study, site specific SI curves must be produced to have any confidence in HSI. Awareness of limiting factors in the original selection of variables is also crucial since the calculation of HSI can heavily weight low or minimum SI scores, which are those which should determine habitat suitability. The range of variables that are selected in the initial scoping process must include potentially limiting factors. Otherwise comparisons of HSI between locations and through a time series could lead to erroneous management decisions.

An alternate approach to using SI curves in the HEP is the application of regression models to determine HSI. In developing models of suitability in reservoirs Aggus and Bivin (1982) defined habitat suitability as a direct measure of fish biomass estimated from harvest data. The assumption here is that the suitability of a reservoir to support fish is reflected in the creel harvest which targets only the older age classes. The method applies to coho, kokanee, cutthroat, rainbow, brook, brown, and lake trout, northern pike, yellow perch, and walleye. Aggus and Bivin used a stepwise regression analysis to select the three independent variables that produced the highest r^2 for all species. A cumulative frequency plot was then developed for use in changing harvest predictions from the regression models into a rating scale (0 to 1) that would be synonymous with HSI. Harvest estimates for a given species were ranked in increasing order. The cumulative frequency was the proportion of harvest estimates that were less than or equal to a harvest estimate.

The advantage of this approach over the use of SI curves is that actual harvest data or that from the pertinent regression model can be used to estimate HSI. There may be less uncertainty than may be present in SI curves, particularly if they must be determined on the basis of expert opinion. A basic assumption, however, is that harvest data give an index of suitability, which means that exploitation must not have reduced population numbers below a level that the habitat is capable of supporting and that harvesting pressure must be the same between systems and times that are being compared. If this is not true or harvesting information is not available, there may be considerable risk in applying the technique. In such a case, the use of SI curves may be more reliable.

In a brief review of trials using HEP in British Columbia, Levy (1993) cited generally poor results. To avoid calculation of HSI every time the HEP is used, the US Fish and Wildlife Service has developed HSI's for several sport and commercially harvested fish (Table 4.1). The HSI models taken from US sources tend to emphasize limitations on suitability by summer habitat. But, suitability of coastal B.C. habitat for coho, for example, is thought to be more limited by winter processes (Hartman and Scrivener 1990). Hence, it was not surprising that no correlation was found between HSI and fall coho abundance measures in work cited by Levy. In tests on the Coldwater River, HSI's developed in the US for coho and chinook were found to contain too many irrelevant variables. Models developed for coastal streams were not considered appropriate for interior systems based on the analyses cited by Levy (1993).

Table 4.1. Listing of available habitat suitability indices for B.C. sportfishes used in HEP

Species	**Reference**
Rainbow trout	Raleigh *et al.* (1984)
Brown trout	Raleigh *et al.* (1986)
Brook trout	Raleigh (1982)
Cutthroat trout	Bovee (1978)
Lake trout	Marcus *et al.* (1984)

Arctic grayling	Hubert *et al.* (1985), Reynolds (1989)
Chinook salmon	Lister (1988), Raleigh *et al.* (1986)
Coho salmon	Lister (1988), McMahon (1983), McMahon (1987)
Chum salmon	Hale *et al.* (1985), McMahon (1987)
Pink salmon	Raleigh and Nelson (1985)
Sockeye salmon	Bovee (1978)
Kokanee	Bovee (1978)
Yellow perch	Krieger *et al.* (1983)
Walleye	McMahon *et al.* (1984)
Northern pike	Inskip (1982)

In any application of HEP to British Columbia lakes and streams it is apparent that preliminary site specific development of SI curves and careful selection of an aggregation method to determine HSI will be required. At the intensive level of management, this *a priori* work will be essential. Assuming that mechanistic methods would be used in determination of HSI, it is clear that SI curves developed in the US can not be extrapolated to B.C. with confidence. New SI curves determined either by experimentation, or from site specific investigations already reported in the literature must be completed before HEP can be applied.

There may, however, be an exception to this requirement. At the reconnaissance level, HEP estimates need not be precise but "reasonable" accuracy is important. Most applications will involve repeated estimates of HSI through time to explore impact hypotheses or effects of mitigation techniques at the basin scale. Hence, consistent sensitivity to relative change is important, which means that consistency in the accuracy of repeated estimates over the range of values described in SI curves is important. If this condition can be met by many of the SI curves already available from work in the US, the development of new curves may not be necessary. However, their use must be restricted to "gaming" or exploratory investigation of "ball park" change in suitable habitat in relation to a stress or mitigation imposed within a drainage basin. Where major shifts in HSI are detected, more detailed work using site specific SI curves to improve accuracy and precision of HEP estimates and establish greater insight into the relative importance of various control variables will be required.

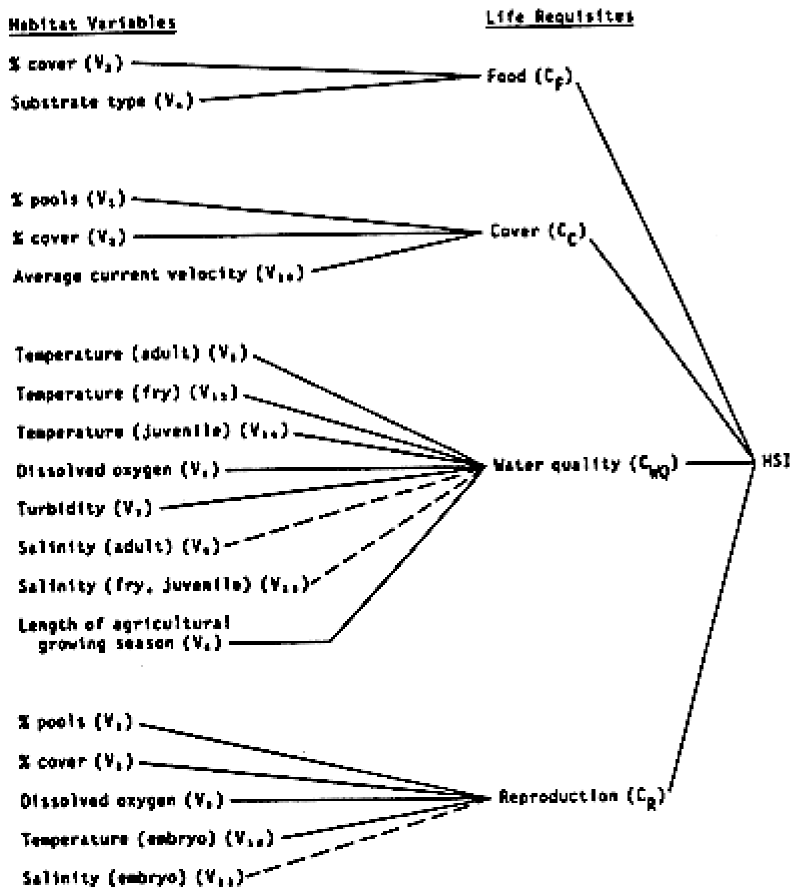

Figure 4.1. A generic mechanistic model used to determine HSI in the Habitat Evaluation Procedure (HEP). Solid lines show how habitat variables are combined to estimate HSI. Dashed lines indicate optional variables. Reproduced from Levy (1993).

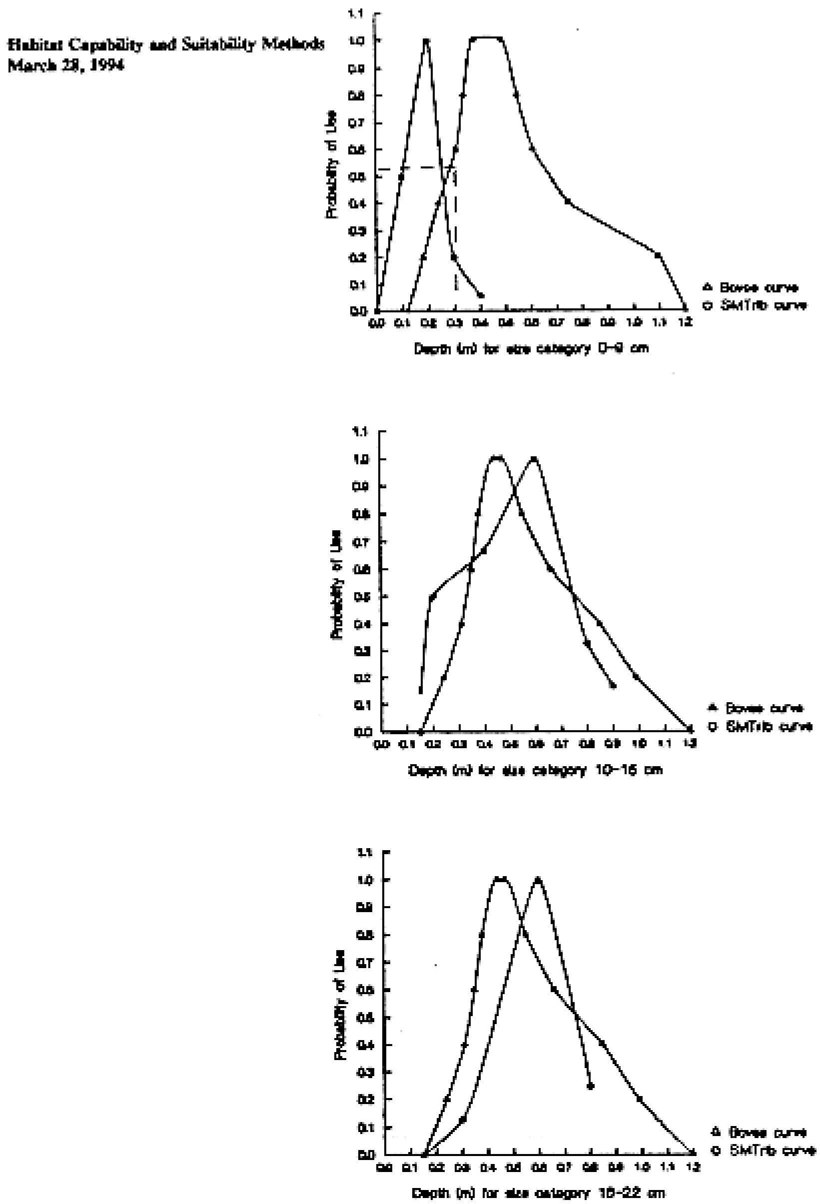

Figure 4.2. A comparison of published and site specific SI curves that can be used to determine HSI in the Habitat Evaluation Procedure and for determination of weighted usable area (WUA) in the IFIM The dashed line indicates how a SI value is determined from an independent microhabitat variable. Reproduced from Oliver (1994).

4.2.2 Instream Flow Incremental Methodology (IFIM)

Detailed descriptions of IFIM are provided by Bovee (1982) and Armour *et al.* (1984). The following summary is taken from these reports.

After the initial scoping is complete, an initial field reconnaissance is conducted to delineate study reaches and select sampling sites. Four features must be identified on maps:

- the confluence of major tributaries and diversions;
- spacial zonation of gradient, fish species distribution, and values of water quality parameters that are observed in the field or measured in more detail;
- delineation of unique habitat types that are essential for any life stage of the target species but are limited in areal extent in the reach(es) potentially influenced by changes in flow; and
- random selection of reaches to be sampled having habitat types that are relatively abundant.

Sampling sites are selected to ensure that data can be extrapolated over the entire project area. Transects are then laid out to characterize the hydraulic microhabitat conditions. Microhabitat refers to the aggregate of the three dimensional distribution of velocity, depth, substrata characteristics (*e.g.*, identification of mud, sand, cobble, bedrock), and cover. Repeated measurements are then taken of the microhabitat variables to document their longitudinal and lateral distribution at several different stream flows. All procedures for data collection are specified in IFIM methods documentation (*e.g.*, NERC 1992) to ensure that the data are compatible with input formats to the computer models that must read that information.

The physical data (velocity, depth, substrata, and cover) are then used by a physical habitat simulation model (PHABSIM) which determines the velocity, depth, substrata, and cover at various flows. If other factors are also thought to change and potentially affect habitat suitability for the target species, associated models can be run with PHABSIM. For example, temperature (Theurer *et al.* 1984) and water quality (Brown and Barnwell (1987) include dissolved oxygen, biological oxygen demand, total ammonia, nitrate, and total phosphorus as water quality descriptors) models that can be linked to PHABSIM are available to simulate effects of variation in flow on those parameters. Output from the simulations separates the study reach into a series of adjoining cells. Any one cell contains homogeneous microhabitat and macrohabitat (variables other than velocity, depth, substrata, and cover) conditions for a given flow.

Suitability curves (Figure 4.3) are then used to determine the suitability of a simulated velocity, depth, substrata, and cover, for a given species and life stage. These SI curves for the microhabitat variables only apply to cells for which there are suitable conditions of temperature, water quality, food production, or any other macrohabitat variable. A fundamental component of PHABSIM is that a suitability index must be assigned to each simulated variable for the species and life stage that is being investigated. This is the same SI that is used to determine HSI in the HEP method. Unlike HEP in which SI curves are selected for any variables that are thought to best represent factors that limit suitable habitat, SI curves in IFIM only pertain to velocity, depth, substrata, and cover. These SI's are then aggregated by multiplication to determine a composite suitability (CSI_i):

$$CSI_i = f_v(V_i) \times f_d(D_i) \times f_s(S_i) \times f_c(C_i)$$

where V = velocity, D = depth, S = substrata, C = cover and f_v, f_d, f_s, and f_c are weighting factors that quantify probability of use as a function of velocity, depth, substrata, and cover respectively within a given cell ($_i$) as determined from the suitability curves. These curves which have become known as probability of use (Bovee and Cochnauer 1977) or suitability index (SI) curves (Bovee 1982) are thought to describe the behavioral characteristics of a life stage of a particular species with respect to each of the microhabitat variables. The peak of the curve gets a weighting value of 1 and represents the optimum value of a variable for use by a given life stage of a given fish species. The tails of the curve get a weighting of 0. Where possible, the curves are empirically determined using instantaneous measurements of the distribution of fish over the range of the variable being examined at carrying capacity and without exploitation. If data are not available, the USFWS describes a procedure known as the "Delphi technique" (Crance 1987) which relies on expert opinion to develop the SI curves. Alternatively Thielke (1985) proposed a logistic model to predict the probability that a fish will occupy a specific location:

$$S(f/n) = \exp(X)/(1 + \exp(X))$$

where S(f/n) is the predicted proportion of fish (f) in sample size n and X is a linear function of independent physical measurements. Using a stepwise regression procedure, X was best determined as a function of categorical values of velocity cover, depth class, and substrate code. Thielke (1985) points out, however, that the approach has only been introduced and without testing and further development at sites other than where it was initially developed, it may not be appropriate for estimating suitability of use. Clearly it should not be used in B.C. until testing suggests it may be a reliable predictor of SI for B.C. species.

The composite suitability index is then multiplied by the area of the cell to obtain the weighted usable area for the cell. The weighted usable area for the study reach (WUA) is the sum of composite indices multiplied by cell area for all cells in the reach:

$$WUA = \sum_{i=1}^{n} (CSI)_i A_i$$

where A_i is the area of cell *i* and *n* is the number of cells in the reach. WUA is determined for each life stage and each flow of interest. Change in WUA as a function of microhabitat can then be examined in a series of two dimensional plots that can be produced for each species and life stage. The lineal extent of an acceptable range of temperatures, water quality conditions, or any other macrohabitat feature that can be modelled and will vary as a function of flow is then determined for given time periods. For example, the temperature of water withdrawn from the hypolimnion of a reservoir will increase as a unit volume of that water is entrained in a downstream river. That hypolimnetic water may also have been characterized by high dissolved phosphorus concentrations, but in the river, the P concentrations are reduced due to uptake processes by the benthic community in the river. If those lineal changes can be modelled or they are known, the lineal extent of acceptable temperatures and P concentrations for usable habitat for a given fish species can be defined for given time period of interest. Separate lineal distances can be determined for every time period of interest. Total habitat area (HA) is then determined as:

$$HA = \text{usable length} \times WUA$$

Effects of flow manipulation or mitigation can then be explored with plots of HA or WUA against flow for specific reaches. Longitudinal profiles of WUA for given flow regimes can be examined. Time series plots that include reach comparisons or implementation of some management action on flow can also be examined. These and several other approaches (NERC 1992) can be used to provide information for optimization of flow regimes for protection of usable habitat and time series changes in usable habitat area with and without flow manipulations or mitigation actions.

Soon after IFIM was formally introduced (Bovee 1982), numerous criticisms appeared in the literature indicating that crucial assumptions of the method could not be upheld in actual practice (Mathur *et al.* 1985, Shirvell 1986, Scott and Shirvell (1987)). There were three main points which are presented below.

SI curves are not probability functions.

The calculation of the composite suitability for fish of a given life stage in a given cell assumes that weighting factors are based on probability distributions. Mathur *et al.* (1985) point out that SI curves are not probability functions but only ratios based on counts of fish relative to a maximum that was encountered on dates of observations. Further, the SI curve may change from one date to the next and from one river to the next, thus introducing considerable uncertainty in determining what is the real "preference of use". Also, a requirement of PHABSIM is that the SI curves are determined from an unexploited population at carrying capacity. Mathur *et al.* (1985) argue that such a condition is rare and usually unknown if modelling approaches must be used to examine functional responses in streams. If reaches that are sampled to determine the curves are not at carrying capacity, the "preference curve" may be flawed and variation in curves determined at different times and locations should be expected. Indeed, Mathur *et al.* (1985) show considerable variation in cited SI curves determined for the same species and life stage and recently, Oliver (1994) also showed considerable variation between published (Bovee 1978) and site specific SI curves for westslope cutthroat trout.

There is no consistent positive linear relationship between WUA and fish abundance.

The biggest criticism is that the assumption of a positive linear relationship between WUA and fish abundance is fundamentally flawed. Although such a relationship has been found for some streams in Colorado (Nehring 1979, in Conder and Annear 1987), Wyoming (Stalnaker 1979, in Conder and Annear 1987), Tennessee and North Carolina (Loar 1984, in Conder and Annear 1987), and New Zealand (Jowett 1992) there many more observations of poor or negative correlations (*e.g.*, Orth and Maughan 1982, Shirvell and Morantz 1983, Irvine *et al.* 1987). Scott and Shirvell (1987) suggest that the frequency of finding a positive linear relationship between WUA and fish abundance or biomass is so low, that it can be due to chance alone. In rebuttal, Orth (1987) stated that the reason a consistent relationship between WUA and fish biomass cannot be measured is that instantaneous counts of fish are related to past habitat limitations for any life stage. Hence, instantaneous estimates of suitability should not be related to instantaneous population size. Orth's article may be viewed as a more detailed explanation of ecological processes to support the initial rebuttal (Orth and Maughan 1986) of the criticism by Mathur *et al.* (1985). Orth clearly stated that users of IFIM must consider factors other than WUA that may limit fish abundance to be successful in applying IFIM concepts. Those other factors may include food deficiency (Irvine *et al.* 1987, Jowett 1992), predator-prey interactions (Bowlby and Roff 1986), "bottleneck effects" between winter and summer habitat (Nickelson *et al.* 1992), and macrohabitat conditions not considered in supplementary IFIM models including turbidity, available spawning area, and others (*e.g.*, Levy 1993).

Component variables of WUA are not independent.

The derivation of WUA assumes independence of component variables (velocity, depth, substrata, cover). However, depth and velocity are known to be related (Mathur *et al.* 1985) which means that the selection of a certain velocity by a fish may not be unrelated to the effect of depth and vice versa. This relationship, however, is largely academic since SI curves used to determine WUA are not probability functions, which, from a statistical viewpoint, means it doesn't matter how the components of WUA interact.

Despite these major criticisms, IFIM continues to be applied to virtually all flow manipulation projects in the US and many in Canada (Reiser *et al.* 1989). Although this fact appears to contradict common sense, another fact is that the Instream Flow Group of USFWS continues to make modifications and attempts to improve on the original

IFIM. They also offer extensive training programs and courses on IFIM and in-stream methodology in general which are advertised and attended widely. To date this exposure has relegated other techniques for estimating suitability to a less prominent position and produced a "popularism" for use of IFIM or parts thereof. Hence, any development of alternate approaches is less well known. The fact that there is considerable pressure in the northwestern US and British Columbia to implement small hydro projects to feed a demanding power grid (M. Henry, Federal Energy Review Commission, Portland, OR. pers comm.; K. Stein, USFS, Portland, OR, pers comm; G. Mathews, B.C. Hydro, Burnaby, B.C. pers comm) has also helped to fuel the use of IFIM. When asked why IFIM continues to be so popular, given its problems, every response is "I know it has problems, but show me something better". This is a reasonable comment that highlights the problem. There is a need for alternative methods but, to date, the infrastructure supporting IFIM appears to preclude any new initiative. There is a "momentum" for use of IFIM that defies logic, is increasing, and excluding development of reasonable alternatives.

Given that IFIM appears to be here to stay, at least for the near future, *it is absolutely critical that it be applied to problems for which it is designed*. It was not designed to provide a definitive answer for application to flow disputes (Armour and Taylor 1991). It was designed to provide a tool to systematically evaluate alternative flow regimes for people involved in stream flow decision processes. The main focus must be on exploring relative change in "potentially usable area" of habitat that can change in relation to manipulation of flows. Whether fish abundance actually does change with shifts in that habitat area is the subject of other study that may accompany IFIM or can be established from the literature or in site specific experimentation. By combining IFIM with other approaches to examine alternative variables that may affect fish abundance, the importance of the inherent assumption of a positive linear relationship between WUA and fish biomass is reduced. Limiting factors that determine fish abundance must be assigned to another process of site specific experimentation, monitoring, or other modelling techniques. This flow of information from IFIM and other sources supports the view of Orth (1987) that IFIM is not a quantitative model; it is a process to assist with decision making. WUA must only be considered a relative index of potentially suitable habitat for fish to use that can be compared in time and space. From this interpretation, WUA should properly be considered an index of weighted potentially useable area.

The misuse of IFIM when applied to large projects can result in large expenditures of time and money involved with dispute resolution, extensive technical debate, media coverage that is suspicious of science, and little contribution to resolving uncertainty in effects of changes in flow on salmonine habitat. A classic example of misuse in British Columbia is the application by the Aluminum Company of Canada (Alcan) to divert flows from the Nechako River through a tunnel to the Kemano power station; known as the Kemano Completion Project (KCP). Various impact assessments were well underway in the early 1980's, at the same time that the Bovee (1982) monograph describing IFIM methods became available. Alcan used the basic concept of IFIM, with modifications of the physical models, to predict the effects of reduced streamflow on the abundance of anadromous salmonines (Shirvell 1987). That objective inferred quantitative predictions that were more demanding than IFIM concepts were capable of providing. Hence, criticism that was already encircling IFIM at the time was easily directed to KCP and now exists as formal expert reporting (Shirvell 1987). Aside from uncertainty with the application of physical models that were used in place of those usually associated with PHABSIM (Shirvell 1987), three general and conceptual criticisms were outlined, all of which pointed to inappropriate use of IFIM or its components.

- The objective assumes a positive linear relationship between WUA and fish abundance but this is not known to exist in the Nechako River. This fact invalidates inference between flow manipulation and salmonine abundance using IFIM.
- PHABSIM cannot make predictions about habitat area after change in flow.
- Abundance of salmonines in the Nechako River may be limited by factors other than microhabitat variables. PHABSIM fails to include other factors and thus cannot be applied to the Nechako River. Food limitation, temperature, and total dissolved gas saturation were variables cited as potentially important but they were not included with PHABSIM simulations.

In another analysis of IFIM data developed for the Nechako, Slaney *et al.* (1984) indicated that in addition to the problems cited by Shirvell (1987), its use was very limited due to sources of error from the influence of anchor and frazil ice formation on physical model predictions, inadequate amount of sampling, and potential alteration of the river channel which was not accommodated in the physical simulation models.

With the introduction of IFIM, a single technique was suddenly available for an environmental assessment that was needed on the Nechako. In retrospect, users appeared to jump at the opportunity to apply a tool that would "answer all the important questions" without detailed consideration that there were very good reasons why such a tool had not been previously available. The approach provided a framework for bias to be an integral part of the analysis. Error can easily be introduced into the outcome of IFIM analyses unless the fundamental details of the functioning of the approach are well understood. In this case, error (although it may have been unintentional) was introduced by not relating intended objectives to the capabilities of IFIM. As cited by Lamb (1989) in his essay on complex system analysis, error and uncertainty can haunt analysts when they overlook "fuzziness", view data as pure, ignore important variables, and accept model outputs as truth. This lack of attention to detail in the use of IFIM has certainly come to haunt Alcan as the company is now faced with extensive construction delays, repeated public hearings, unnecessary costs, and a degree of uncertainty in the fate of the KCP. This example should be a warning to anyone interested in the application of the IFIM process to any water manipulation project to be absolutely certain what IFIM is capable of providing and not to push it any further.

An example of testing assumptions of IFIM before its potential use was in a study of factors determining brown trout abundance in New Zealand streams by Jowett (1992). Multiple regression models of brown trout abundance were developed from independent variables describing hydrology, temperature, physical characteristics, weighted usable area (based on IFIM criteria), fish food organism biomass, chemistry, and catchment characteristics. In selecting variables that best explained variability in brown trout abundance, WUA combined with food abundance were found to be the most important variables. A significant linear relationship between WUA and trout abundance was confirmed. The SI curves that showed the best correlation with trout abundance were those developed in Brown trout streams of New Zealand, thus indicating the value of developing site specific SI before determination of WUA. Jowett also found that WUA for food production (following methods described by Waters, 1976) was an important determinant of fish abundance. Consideration of factors other than a simple relationship between fish abundance and habitat area that may limit fish abundance was thus applied, and the regression models showed that food production plus space was crucial. This *a priori* assessment thus dealt with the most important criticism of IFIM (that habitat area and fish biomass may not be related). Assuming there is confidence with the physical modelling in PHABSIM, output from an IFIM application in these New Zealand streams could be related to trout abundance. A similar approach will be required in B.C. if IFIM is to applied to projects where a quantitative relationship between WUA and fish abundance is required. It requires a "bundled" approach of IFIM with quantitative analyses of empirical data or experiments to establish relationships between fish abundance and indices of suitability.

4.2.3 Missouri Stream Habitat Evaluation Procedure (SHEP)

A detailed description of SHEP is provided by Fajen and Wehnes (1981). It applies only to streams. The following summary is taken from this report.

SHEP assigns a rating (0-10) to the quality of fish habitat for 6 parameters.

1. Barriers to fish migration. Barriers are considered dams or other structures that produce vertical drops. The rating applies to the relative amount of vertical drop; 10 is no obstruction, zero is several obstructions causing a drop of more than 3m at low flow. Data can be collected from maps or in field observations.

2. Percent of watershed in urban development. Development of communities are inferred to reduce the quality of streams. A 10 is <5% of the watershed is in urban development; zero is >70% is in urban development. Information is determined from maps and air photos.

3. Condition of riparian vegetation. The riparian zone is recognized for moderating water temperature, providing protection from adjacent land use activities, and providing cover for fish. A 10 means that riparian vegetation covers all stream banks in the study reach. Zero means <10% has riparian vegetation. The rating is determined from air photos and on-site observations of vegetation cover within 30m of the stream bank.

4. Degree of erosion in the flood plain. This is mainly a rating of channel stability and potential to produce turbidity. A 10 means there is no evidence of erosion. Zero means the floodplain is severely eroded and the stream channel is poorly defined. The rating is determined from air photos, on-site observations, and reference to soils data if available.

5. Percent protected from land use impacts. Soil and vegetation conservation practices are recognized

to influence water quality, substrata characteristics, flow variation, and potential for erosion. A 10 means that the watershed has >80% undisturbed forest, improved pasture terraces, or other conservation practice. Zero is for no mitigation and extreme land disturbance. The rating is determined from air photos and on-site observations.

6. Flow alteration. This parameter mainly applies to the number of flow control structures that may influence the natural stream hydrograph. These include ponds with flow control structures and other impoundments used for irrigation or domestic water supply. A 10 means that <10% of the watershed is controlled by impoundments. Zero means that >95% of water flow in the watershed is controlled by impoundments.

In addition to these habitat quality parameters, SHEP uses 4 habitat alteration functions which act to alter the habitat quality indices depending on the type and extent of a habitat alteration. Channel modification rate (f_1) is used to adjust suitability based on an arbitrary assignment of percent fish reduction associated with removal of riparian vegetation (25% fish reduction), intentional alteration of channel geomorphology (80% fish reduction), and changing a natural channel to a paved conduit (95% fish reduction). Channel modification rate is determined as:

$$f_1 = 1.0 - (\%\ \text{modification} \times \%\ \text{fish reduction})$$

Impoundment rate (f_2) assigns an arbitrary type of degradation (0% is no impoundment; 100% is stream is fully impounded for water conservation) associated with habitat loss with the amount of impoundment and is determined as:

$$f_2 = 1 - (\%\ \text{degradation} \times \text{rating of type of degradation})$$

Water quality (f_3) is simply another rating from pristine water (rating of 1) to polluted water that causes fish kills (rating of zero). No input parameters or measurements are required. This is only a subjective scale that must be based on expert opinion.

Streambed condition (f_4) is still another rating extending from stable substrata (rating of 1) to a reach filled with unconsolidated fines and unstable material (rating of zero).

A habitat index is then determined as:

$$HI = (\Sigma P_i / N_p) \times f_1 \times f_2 \times f_3 \times f_4$$

where P_i is the individual parameter values and N_p is the number of parameters used.

HI varies from 0 to 10 and gives a relative index of "habitat quality". It has several shortcomings that make it inappropriate for use in British Columbia. The index has only been used in catfish and bass streams of Missouri. It has never been tested on streams that support sportfish found in B.C. and thus requires validation. HI is an arbitrary index based on ratings scales that are largely subjective and can be biased by opinion. Hence, the system lacks any basis, statistical or otherwise, for space and time comparisons which are essential for suitability estimates in B.C. The rating scales always assume linear associations between any disturbance and the suitability of habitat for fish but the authors provide no supportive evidence, even for the sites from which the method was developed. Hence, any relationship between HI and fish habitat characteristics is unknown. It is also unknown if HI has any relevance to the suitability of habitat for fish in British Columbia. We are not aware of the method being applied outside of Missouri. Although a similar rating scheme may be considered for fish species in specific regions of British Columbia, the basic problem of user bias, lack of comparability, and uncertainty of the relevance of HI makes the approach inappropriate at this time.

4.2.4 Fish Habitat Index (FHI)

A preliminary description of FHI is provided by Parsons *et al.* (1981). It applies only to streams in forested watersheds with and without forest harvesting activities. The following summary is taken from that article.

FHI is derived by multiplying habitat area by habitat quality (a numerical indicator of habitat condition). FHI is determined in both undisturbed and harvested conditions. Area is the area of fish bearing streams which is assumed to decrease with harvesting activities due to stream blockages from log jams. Habitat quality is a rating from 0 to 10 that is determined using a regression equation that is based on geomorphic parameters. This quality is assumed to decrease in harvested drainage due to sedimentation, increases in water temperature, debris torrents, slope failures, etc.

The regression model for undisturbed watersheds was developed by the authors to predict the quality value using four independent geomorphic parameters:

> Quality Value = 6.56 + 1.44(basin perimeter) + 0.00089(basin relief) - 2.02(basin area) - 5.62 (compactness coefficient).

Total FHI for a landtype association is determined by multiplying the quality value from the regression model by area of landtype. The authors indicate that separate regressions for areas disturbed by forest harvesting activities have been developed but these were not included in the article.

The approach may be appealing for overview techniques if the geomorphological data were available. It is simple to use and can be compared spatially and temporally. However, the model has not been used except in the location where it was developed and thus it has not been tested and verified for use elsewhere. Consequently the model is not recommended for use in B.C. at this time. It may, however, be considered if any new model development is planned at the overview level of management. It is recommended that the developers of the model be contacted to determine if the model is still in use and if so, find out if it has changed and verified at other locations. If so, it may be worth testing in forested watersheds in B.C.

4.2.5 Planned Reservoir HIS

An index used to rate the suitability of planned reservoirs for rainbow trout and yellow perch was developed by McConnell *et al.* (1984). The following summary is taken from the McConnell paper.

Habitat suitability is determined from rating 5 main "attributes" including temperature, mineral turbidity, structural cover, maximum drawdown and timing of drawdown, and frequency of shallow coves. Each primary attribute is scored within one or more matrices that include cross comparisons of secondary attributes that require input of data. For example a temperature score for rainbow trout requires information on three secondary attributes (why three are determined and not others is not described) including a climate score (matrix of mean July air temperature versus length of growing season), operations score (a matrix of storage ratio versus depth of outlet), and stratification score (a matrix of maximum fetch versus mean depth). There are three ratings for each variable on the matrix borders, resulting in a matrix for each secondary attribute containing nine squares that represent cross comparisons in the matrix. Within each box a rating of 1,2, or 3 is assigned on the basis of local knowledge or opinion of the interactions of boxes within the matrix. Data for the matrices can be available climate data, and engineering plans for the reservoir. The 1, 2, or 3 code for each of the 5 main attributes is then compiled to form a 5 digit code. A series of tables are provided which give a suitability rating for all combinations of the 5 digit code.

Although this system is appealing for overview estimates of suitability, it is entirely based on subjective ratings. Hence the approach has no basis for spatial and temporal comparisons. Although simplistic, it is not clear in the evidence presented by McConnell *et al.* (1984) why certain attributes were selected for a given species. These may also have been subjective assignments. Given these uncertainties, the inability to reliably use the method for site and time course comparisons, and that it can only be applied to sites for reservoirs that presently do not exist, it is not recommended for use in British Columbia.

4.2.6 Tennant flow method

The Tennant method is very simplistic and has received appeal in regions of British Columbia as a quick and easy-to-understand method of determining habitat and recreational suitability of stream habitat. The method is based on the assumption that flows that are satisfactory for needs of fish and other aquatic biota will also be sufficient for maintaining recreational and aesthetic qualities. Based upon empirical relationships and observations, Tennant suggests that minimum flows at any time of the year must be >10% of mean annual discharge. Below the 10% threshold, fish habitat and recreational value will be severely degraded. Above 10%, habitat and recreational quality increases in a range from fair conditions (10% in winter and 30% in summer) to outstanding (40% in winter and 60% in summer). An optimum range is considered 60-100% of mean annual flow at any time of the year.

This approach is very appealing due to its simplicity and for this reason it has been applied in British Columbia. Many regional fisheries biologists have used it for obtaining rough estimates of suitability where the flow data are available. Where validated, it is appropriate for use at the overview and reconnaissance levels of management. However, we are not aware of projects in which it has been validated. Hence, the collection of observational or ideally experimental data similar to that compiled by Tennant (1975) from sites in warm and cold water streams of the US midwest, great plains, and western mountain ranges are required for validation purposes before the method can be recommended for routine use in British Columbia. The method is further limited by the availability of annual flow records. In British Columbia, however, the province-wide network of flow monitoring sites managed by the Water Survey of Canada may provide adequate records for sites in close proximity to a gauged location.

4.3 Methods Recommendations

Of the six suitability methods outlined in Section 4.2, only IFIM can be immediately used in British Columbia, but only in large, big budget projects requiring an intensive method to examine relative change in habitat suitability with respect to a flow manipulation. IFIM can be used as a tool to systematically evaluate alternative flow regimes for people involved in stream flow decision processes at the reconnaissance and intensive levels of management. It is not a quantitative tool used to provide a definitive answer to flow disputes. To develop confidence and reliability in IFIM, the following recommendations must be followed:

- recognize that IFIM is a "process" to assist with decision making. It does not produce decisions and it does not provide data to quantify change in fish abundance as a function of flow manipulation;
- there must be some evidence either in empirical data or literature sources that the abundance of a life stage of the fish that is being examined is not limited by macrohabitat conditions (*e.g.*, food production, predator-prey interactions, temperature, etc.). This is very difficult to resolve unequivocally without extensive experimentation or data collection (*e.g.*, Jowett 1992) but at very least, some effort must go into this analysis to set "reasonable" spacial and temporal bounds first on the determination of WUA and then on the extrapolation to HU;

- where possible, site specific data should be collected to examine the relationship between fish abundance and WUA. If it is not linear, WUA can only be considered an index of potential suitability and it cannot be directly related to fish abundance;
- site specific SI curves should be determined for the study area; and
- ensure that there are no confounding factors (*e.g.*, ice formations) that may interfere with the ability of PHABSIM to correctly predict change in microhabitat variables with varying flow. If confounding factors are present, models that are sensitive to the effect of all factors should be used in place of PHABSIM or the entire process should not be used.

These recommendations require considerable effort on behalf of potential users of IFIM even before the first output of WUA can be used. For example, Oliver (1994) suggested that sampling effort required to obtain reasonable precision in determining SI curves was "unrealistic for most management purposes" and due to inherent variability between habitats within a stream reach, IFIM procedures should be restricted to assessments in discrete habitat units, not whole streams or whole reaches. Given this constraint, IFIM is not a technique that can be run quickly. It is suitable only as part of detailed analyses of large projects that require detailed investigations.

HEP has been tested at a few sites in British Columbia will poor results (Levy 1993) due to inappropriate selection of habitat variables and a general lack of site specific SI curves. The biggest problem with HEP is that it requires extensive *a priori* knowledge of factors that limit fish abundance at a given site. This requirement is much more important for HEP than for IFIM since variables must be selected for HEP that actually do limit fish abundance. As pointed out by Bisson (1992), an unequivocal determination of limiting factors requires extensive experimentation. If this research effort is required, it is likely that the suitability of habitat to support fish will be determined long before one is ready to apply HEP. This requirement for substantial insight into limiting factors is the fundamental problem with HEP, regardless of other difficulties in providing reliable SI curves and an appropriate method of aggregating SI data to determine the habitat suitability index. We regard these problems intractable for present or future application of the method in British Columbia and thus HEP should not be considered for any application in the province. This finding is consistent with recent evaluations in the US where HEP is also falling out of favour for use in assessing fish habitat suitability (K. Fausch, Colorado State University, Pers. Comm).

Of the simpler approaches described in section 4.2, the Tennant method has been picked up as a quick approach for estimating suitability in various regions of the province. However, it has never been formally validated and this testing must be completed before it can be recommended for routine use.

Each of SHEP, FHI, and the planned reservoir HSI were developed primarily as concepts that the authors have indicated require further research and testing before they can be applied elsewhere with confidence. These methods have never been applied to sites in British Columbia and as part of a long term strategy for methods development, they should be considered for additional testing. However, they do have problems other than the lack of validation that must be considered as part of any testing initiative. As discussed in Section 4.2.3 SHEP:

- has not been tested using species that are found in B.C.;
- it is subject to bias in the assignment of rating scores;
- lacks statistical rigor for any time and space comparisons; and
- was developed only for agricultural areas having impacts from urban development and stream channelization.

However, the basic concept of assigning quick rating scores is appealing for application to overview and reconnaissance applications and thus we recommend this method be considered as part of methods development and validation. Existing parameters in the method will likely have to be changed, tested, and verified to reflect conditions of B.C. waters. The quantitative scoring system used in the fish habitat index may be suitable for spatial and temporal comparisons and is also appropriate for further testing and verification. Resolution is at the basin or watershed scale, making it suitable for overview and reconnaissance level estimates of suitability. The planned reservoir HSI may have big problems with user bias and appears to lack a clear process for the selection of "attributes" in applications remote from where it was developed. However, the concept is novel and it should not be discarded outright from future testing. With work, the process may be standardized to minimize potential bias and provide a tool for exploring change in habitat suitability with various scenarios of reservoir size and operational strategy.

Our analysis indicates there is no readily available "off the shelf" method to estimate habitat suitability quickly, on a low budget, in any habitat and at any management level. When these estimates are required, which is likely to be most of the time, capability estimates will presently have to be substituted, where possible. This recommendation is based on the view that if an area is capable of supporting fish, then the habitat is also suitable as fish habitat. The reverse may not be true. Recommended methods of estimating capability for all management levels are provided in Section 2.0.

Given the limited availability of suitability models, it is recommended that a program of verification and testing that is required for most existing models also consider the development of new methods for use in British Columbia. There are at least two and probably more concepts to consider in this process. One is to develop a rating of habitat that is *unsuitable* for the growth of fish in early life stages. There are volumes of empirical relationships in the literature describing physiological and pathological responses of organisms or populations to chemical and physical manipulation of lake and stream habitat. It may be easier to apply these findings to rank habitat according to tolerance levels by lake and stream biota. This approach is the basis of toxicity guidelines in both the US and Canada (*e.g.*, CCREM 1993) and it may be appropriate for ranking habitat suitability as well. A comprehensive approach may be to combine CCREM criteria (which provides threshold data for all chemical and some physical variables), with a ranking for flow (*e.g.*, Tennant 1975), and habitat complexity. Where forest harvesting practices may be producing fines in stream substrata, estimates of embeddedness (*e.g.*, MacDonald *et al.* 1991) may also be considered in a ranking scheme to increase sensitivity to forest management practices. Another approach may be to develop a decision tree that helps determine if a habitat is suitable to support fish. Discussion with regional biologists as part of this project revealed that a standard series of questions are often asked with respect to lake suitability for stocked fish:

- Are seasonal changes in the profile of dissolved oxygen and temperature within the range that can be tolerated by fish?
- Is toxicity from any contaminants an issue?
- Is there a history of winter kill?
- Are there adequate spawning areas?
- Are there abundant fish food organisms present?

Questions pertaining to stream habitat might include:

- Is there adequate cover in rearing habitat?
- Is adequate spawning area present?
- Are summer temperatures and low flow adequate to support fish?

Each of these questions require presence/absence or yes/no type answers and can be resolved with limited field work and reference to standards that are either presently available (*e.g.*, SEP biostandards in the Fish Habitat Enhancement Manual (Adams and Whyte 1990) or can be developed from existing literature. Answers to these questions structured in a decision tree could provide a relatively rapid yet comprehensive approach to assessing habitat suitability. An approach like this may, in fact, be informally already in use in many regions of the province. What is now required is a process to formalize these methods for routine use.

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APPENDIX A DETAILED LISTING OF EMPIRICAL HABITAT CAPABILITY MODELS

LAKE MODELS used for RECREATIONAL/REGIONAL PLANNING, where coarse estimates of stock capacity are estimated from map-based or easily obtained information.

Author	Date	Model #	Spp.	Variables					Model Use Category	R2	DF	R2*DF	N	Terms	Man. Level	Comments on Rank	Comments on Report	Related Models	Data Avail.	Area Description
				dependent	ind. 1	ind. 2	ind. 3	ind. 4												
Jenkins	1982	15.4	mixed	SC	MEI				RP	0.35	287	100.5	290	3	O,R	Rejected: south eastern US reservoirs not that relevant to BC.	Uses, MEI, but polynomial model form rather than linear as Ryder uses. More detail available on # of samples/yr. Also repeated analysis by splitting database into four reservoir types (hydro mainstream, hydro storage, nonhydro HCO3, nonhydro SO4-CL.	12, 34	y	Half of reservoirs in 8 states. Most lie south of 38o N and east of 102oW
Jenkins & Morais	1971	34.5	mixed, RBT	H	effort (hrs/ha)				RP	0.77	101	77.8	103	2	O,R,I	Rejected: if you know effort, you will probably know harvest. No physical predictor.			y	reservoirs throughout US
Godbout & Peters	1988	1.1	BKT	H	mean weight	effort	altitutde		RP	0.92	66	60.4	71	5	R,I	Rejected: if you know mean weight, you probably have some idea of biomass.	All variables are transformed to their common logarithms (base 10).	8,3, 10, 9	y	Laurentian Shield lakes
Jenkins	1982	15.2	mixed	H	MEI				RP	0.21	287	60.3	290	3	O,R	Rejected: south eastern US reservoirs not that relevant to BC.	Uses, MEI, but polynomial model form rather than linear as Ryder uses. More detail available on # of samples/yr. Also repeated analysis by splitting database into four reservoir types (hydro mainstream, hydro storage, nonhydro HCO3, nonhydro SO4-CL	12, 34	y	Half of reservoirs in 8 states. Most lie south of 38o N and east of 102oW
Godbout & Peters	1988	1.4	BKT	H	altitude	effort	area		RP	0.91	66	59.9	71	5	R,I	Rejected: altitude range not applicable to BC.	All variables are transformed to their common logarithms (base 10). Stable catch is defined by the absence of a sign. temporal trend in fishing success for at least 5 consecutive years.	8,3, 10, 9	y	Laurentian Shield lakes
Godbout & Peters	1988	1.3	BKT	H	mean weight	effort	area	TP	RP	0.89	66	58.5	71	5	R,I	Rejected: if you know mean weight, you probably have some idea of biomass.	All variables are transformed to their common logarithms (base 10).	8,3, 10, 9	y	Laurentian Shield lakes
Godbout & Peters	1988	1.2	BKT	H	mean weight	effort	area	TP	RP	0.89	65	58.1	71	6	R,I	Rejected: if you know mean weight, you probably have some idea of biomass.	All variables are transformed to their common logarithms (base 10).	8,3, 10, 9	y	Laurentian Shield lakes
Godbout & Peters	1988	1.5	BKT	H	TP	effort	area		RP	0.88	66	58.0	71	5	R,I	Selected: better indicator for BC than altitude based Godbout paper since alt will be outside of range in BC relative to PQ.	All variables are transformed to their common logarithms (base 10). Stable catch is defined by the absence of a sign. temporal trend in fishing success for at least 5 consecutive years.	8,3, 10, 9	y	Laurentian Shield lakes
Jenkins & Morais	1971	34.2	mixed, RBT	SC	MEI				RP	0.33	137	45.2	140	3	O,R,I	Rejected: south eastern US reservoirs not that relevant to BC.			y	reservoirs throughout US
Scarborough & Peters.	unpbl.	201.5	mixed	H	effort	TP-Area	depth		RP	0.86	42	36.1	46	4	O	Rejected: probably colinearity between TP-Area and depth.	Unpublished manuscript provided by E. Parkinson. Paper also compares Hanson and Legget's, Gobout and Peters' and Ryder's model to theirs using their data set. Data ranges not provided for all lakes.	1, 3, 8, 12,33,	y	Lakes of various trophic levels lying on both igneous and sedimentary drainages in Ontario.

LAKE MODELS used for RECREATIONAL/REGIONAL PLANNING, where coarse estimates of stock capacity are estimated from map-based or easily obtained information.

Author	Date	Model #	Spp.	Variables: depend-ent	Ind. 1	Ind. 2	Ind. 3	Ind. 4	Model Use Category	R2	DF	R2*DF	N	Terms	Man. Level	Comments on Rank	Comments on Report	Related Models	Data Avail.	Area Description
Scarborough & Peters.	unpbl.	201.1	mixed	H	effort	TP	depth		RP	0.85	42	35.7	48	4	O	Selected: better than previous function (201.1) since area removed.	Unpublished manuscript provided by E. Parkinson. Paper also compares Hanson and Legget's, Gobout and Peters' and Ryder's model to theirs using their data set. Data ranges not provided for all lakes.	1, 3, 8, 12,33,	y	Lakes of various trophic levels lying on both igneous and sedimentary drainages in Ontario.
Godbout & Peters	1988	1.8	BKT	H	effort	altitude	chl a	mean weight	RP	0.96	37	35.8	42	5	R,I	Rejected: altitude range not applicable to BC.	All variables are transformed to their common logarithms (base 10). Stable catch is defined by the absence of a sign. temporal trend in fishing success for at least 5 consecutive years.	8,3, 10, 9	y	Laurentian Shield lakes
Godbout & Peters	1988	1.9	BKT	H	effort	altitude	chl a	mean weight	RP	0.96	37	35.5	42	5	R,I	Rejected: altitude range not applicable to BC.	All variables are transformed to their common logarithms (base 10). Stable catch is defined by the absence of a sign. temporal trend in fishing success for at least 5 consecutive years.	8,3, 10, 9	y	Laurentian Shield lakes
Scarborough & Peters.	unpbl.	201.2	mixed	H	effort	TP			RP	0.81	43	34.8	48	3	O	Selected: similar to 201.1 but w/o depth, will more likely to be able to apply it.	Unpublished manuscript provided by E. Parkinson. Paper also compares Hanson and Leggets, Gobout and Peters and Ryder's model to theirs using their data set. Data ranges not provided for all lakes.	1, 3, 8, 12,33,	y	Lakes of various trophic levels lying on both igneous and sedimentary drainages in Ontario.
Scarborough & Peters.	unpbl.	201.4	mixed	H	effort	TP	area		RP	0.82	42	34.4	48	4	O		Unpublished manuscript provided by E. Parkinson. Paper also compares Hanson and Legget's, Gobout and Peters' and Ryder's model to theirs using their data set. Data ranges not provided for all lakes.	1, 3, 8, 12,33,	y	Lakes of various trophic levels lying on both igneous and sedimentary drainages in Ontario.
Scarborough & Peters.	unpbl.	201.3	mixed	H	effort	MEI			RP	0.78	43	33.5	48	3	O		Unpublished manuscript provided by E. Parkinson. Paper also compares Hanson and Legget's, Gobout and Peters' and Ryder's model to theirs using their data set. Data ranges not provided for all lakes.	1, 3, 8, 12,33,	y	Lakes of various trophic levels lying on both igneous and sedimentary drainages in Ontario.
Godbout & Peters	1988	1.11	BKT	H	effort	color (platinum)	chl a	mean weight	RP	0.95	35	33.1	40	5	R,I		All variables are transformed to their common logarithms (base 10). Stable catch is defined by the absence of a sign. temporal trend in fishing success for at least 5 consecutive years.	8,3, 10, 9	n- color	Laurentian Shield lakes
Godbout & Peters	1988	1.7	BKT	H	effort	color (platinum)	chl a	mean weight	RP	0.95	35	33.1	40	5	R,I		All variables are transformed to their common logarithms (base 10). Stable catch is defined by the absence of a sign. temporal trend in fishing success for at least 5 consecutive years.	8,3, 10, 9	n- color	Laurentian Shield lakes

LAKE MODELS used for RECREATIONAL/REGIONAL PLANNING, where coarse estimates of stock capacity are estimated from map-based or easily obtained information.

Author	Date	Model #	Spp.	Variables: depend-ent	Variables: ind. 1	Variables: ind. 2	Variables: ind. 3	Variables: ind. 4	Model Use Category	R2	DF	R2*DF	N	Terms	Man. Level	Comments on Rank	Comments on Report	Related Models	Data Avail.	Area Description
Godbout & Peters	1988	1.12	BKT	H	effort	area	reflectance (%)	effort variance	RP	0.88	37	32.4	42	5	R,I		All variables are transformed to their common logarithms (base 10). Stable catch is defined by the absence of a sign. temporal trend in fishing success for at least 5 consecutive years.	8,3, 10, 9	n-reflectance	Laurentian Shield lakes
Godbout & Peters	1988	1.8	BKT	H	effort	area	reflectance	mean weight	RP	0.89	36	32.1	42	6	R,I		Effort variance (range: ?) is the 5th ind. var. of this model. All variables are transformed to their common logarithms (base 10). Stable catch is defined by the absence of a sign. temporal trend in fishing success for at least 5 consecutive years.	8,3, 10, 9	n-reflectance	Laurentian Shield lakes
Aggus & Bivin	1982	156.1	Wall	H	frost free days	shore develop.	storage ratio		RP	0.32	85	26.8	89	4	O,R	Only applicable to reservoirs.			y	US
Jenkins & Morais	1971	34.4	mixed, RBT	H	MEI				RP	0.29	94	26.3	97	3	O,R,I				y	reservoirs throughout US
Youngs & Heimbuch	1982	21.1	mixed	H	area				RP	0.94	25	23.5	27	2	O,R		Data set compiled from the following reports: Ryder (1965); Oglesby (1977); and Matuszek (1978).		y	
Jenkins	1982	15.1	mixed	H	MEI				RP	0.08	287	23.0	290	3	O,R		Uses, MEI, but polynomial model form rather than linear as Ryder uses. More detail available on # of samples/yr. Also repeated analysis by splitting database into four reservoir types (hydro mainstream, hydro storage, nonhydro HCO3, nonhydro SO4-CL	12, 34	y	Half of reservoirs in 8 states. Most lie south of 38o N and east of 102oW
Youngs & Heimbuch	1982	21.2	mixed	H	area	TDS			RP	0.95	24	22.8	27	3	O,R		Data set compiled from the following reports: Ryder (1965); Oglesby (1977); and Matuszek (1978).		y	
Aggus & Bivin	1982	156.2	NP	H	frost free days	outlet depth			RP	0.67	34	22.8	37	3	O,R	Only applicable to reservoirs.			y	US
Youngs & Heimbuch	1982	21.3	mixed	H	area	TDS	mean depth		RP	0.97	23	22.3	27	4	O,R		Data set compiled from the following reports: Ryder (1965); Oglesby (1977); and Matuszek (1978).		y	
Jenkins & Morais	1971	34.3	mixed, RBT	H	MEI				RP	0.16	137	21.9	140	3	O,R,I				y	reservoirs throughout US
Jones & Hoyer	1982	4.1	mixed	H	chl a				RP	0.83	23	19.0	25	2	R				y	lakes and reservoirs in Mo. and Iowa
Aggus & Bivin	1982	156.3	T	H	surface area	shore develop.	mean depth		RP	0.36	52	18.7	56	4	O,R				y	US
Aggus & Bivin	1982	156.4	RBT	H	surface area	shore develop.	mean depth		RP	0.37	46	17.1	50	4	O,R				y	US
Jenkins	1982	15.3	NP	H	growing season (days)	mean depth (m)			RP	0.53	32	17.0	35	3	O,R		Uses, MEI, but polynomial model form rather than linear as Ryder uses. More detail available on # of samples/yr. Also repeated analysis by splitting database into four reservoir types (hydro mainstream, hydro storage, nonhydro HCO3, nonhydro SO4-CL	12, 34	y	Half of reservoirs in 8 states. Most lie south of 38o N and east of 102oW

LAKE MODELS used for RECREATIONAL/REGIONAL PLANNING, where coarse estimates of stock capacity are estimated from map-based or easily obtained information.

Author	Date	Model #	Spp.	Variables: dependent	Variables: ind. 1	Variables: ind. 2	Variables: ind. 3	Variables: ind. 4	Model Use Category	R2	DF	R2*DF	N	Terms	Man. Level	Comments on Rank	Comments on Report	Related Models	Data Avail.	Area Description
Jenkins & Morais	1971	34.6	mixed, RBT	H	area (ha)	growing season	res. age		RP	0.17	99	16.8	103	4	O,R,I				y	reservoirs throughout US
Hanson & Legett	1982	3.3	mixed	H	TP				RP	0.87	19	16.5	21	2	R,I		Variable ranges pertain to both datasets 1 AND 2 which were used to develop models (3.1, 3.2) and (3.3, 3.4) respectively. Only the most significant SR and MR models reported here. See paper for more. See models 1 (p. 1774) and 168 for validation info.	1, 168	y	North Temperate lakes, 42-62N, 17-117E
Hanson & Legett	1982	3.4	mixed	H	TP	mean depth	TDS		RP	0.97	17	16.5	21	4	R,I		Variable ranges pertain to both datasets 1 AND 2 which were used to develop models (3.1, 3.2) and (3.3, 3.4) respectively. Only the most significant SR and MR models reported here. See paper for more. See models 1 (p. 1774) and 168 for validation info.	1, 168	y	North Temperate lakes, 42-62N, 17-117E
Aggus & Bivin	1982	156.11	T	H	elevation	mean depth	frost free days		RP	0.45	35	15.8	39	4	O,R				y	US
Ryder	1965	8.2	mixed	P	MEI				RP	0.74	21	15.5	23	2	O,R		Commonly cited classic. Includes commercial harvest. See models 1 (p. 1774), 56 (p.175) and 168 for validation info.	1, 168	y	N. Temp. region, alt. < 2000 ft., lake area = 1-31,820 sq. mi. intensively fished
Ryder	1965	8.3	mixed	P	TDS	mean depth			RP	0.76	20	15.1	23	3	O,R		Commonly cited classic. Includes commercial harvest. See models 1 (p. 1774), 56 (p.175) and 168 for validation info.	1, 168	y	N. Temp. region, alt. < 2000 ft., lake area = 1-31,820 sq. mi., intensively fished
Hanson & Legett	1982	3.5	mixed	H	macro - benthos	mean depth			RP	0.83	18	14.9	20	2	R,I		Var1/ Var2 in mod. Var. ranges pertain to both datasets 3 AND 4 which were used to dev. # (3.5, 3.6) and (3.7) respectively. Only most sign. SR and MR models reported here. See paper for more. See models 1 (p. 1774) and 168 for validation info.	1, 168	n	lakes and ponds 0-56N, 121-122W
Ryder	1965	8.1	mixed	P	mean depth				RP	0.69	21	14.5	23	2	O,R		Commonly cited classic. Includes commercial harvest. See models 1 (p. 1774), 56 (p.175) and 168 for validation info.	1, 168	y	N. Temp. region, alt. < 2000 ft., lake area = 1-31,820 sq. mi., intensively fished
Ryder	1965	8.5	mixed	P	TDS	mean depth			RP	0.46	31	14.3	34	3	O,R		Commonly cited classic. Includes commercial harvest. See models 1 (p. 1774), 56 (p.175) and 168 for validation info.	1, 168	y	N. Temp. region, alt. < 2000 ft., lake area = 1-31,820 sq. mi., intensively & lightly fished
Oglesby	1977	6.1	mixed	H	chl a				RI	0.84	17	14.3	19	2	R		Yield expressed as dry wt./m2 of lake surface or as carbon/m2 of lake surface. Variable ranges reported for entire set of 46 lakes only. It may be possible to extract subset ranges if effort is warranted.		y	Data compiled from publ. literature; regions were not defined. Lake location ranged from 10 - 59N.

LAKE MODELS used for RECREATIONAL/REGIONAL PLANNING, where coarse estimates of stock capacity are estimated from map-based or easily obtained information.

Author	Date	Model #	Spp.	Variables: depend-ent	Variables: Ind. 1	Variables: Ind. 2	Variables: Ind. 3	Variables: Ind. 4	Model Use Category	R2	DF	R2*DF	N	Terms	Man. Level	Comments on Rank	Comments on Report	Related Models	Data Avail.	Area Description
Hanson & Legett	1982	3.6	mixed	H	macro - benthos	mean depth	lake surface area		RP	0.83	17	14.1	20	3	R,I		Var1/ Var2 in mod. Var. ranges pertain to both datasets 3 AND 4 which were used to dev. # (3.5, 3.6) and (3.7) respectively. Only most sign. SR and MR models reported here. See paper for more. See models 1 (p. 1774) and 168 for validation info.	1, 168	n- macro - benthos	lakes and ponds 0-58N, 121-122W
Hanson & Legett	1982	3.2	mixed	H	macro - benthos	mean depth	TDS	lake surface area	RP	0.66	21	13.9	26	5	R,I		Variable ranges pertain to both datasets 1 AND 2 which were used to develop models (3.1, 3.2) and (3.3, 3.4) respectively. Only the most significant SR and MR models reported here. See paper for more. See models 1 (p. 1774) and 168 for validation info.	1, 168	n- macro - benthos	North Temperate lakes, 42-62N, 17-117E
Downing, Plante, & Lalonde	1990	42.1	mixed	P	phytoplan kton prod,n				RP	0.79	17	13.4	19	2	R		Data collected from various published sources, so some details not included in paper (i.e. how many seasons/years the data span). PAGE MISSING From Aqu. Res. copy. I relied on their reporting of equations.		n- phytoplan kton prod,n	wide range of geographic areas and trophic status
Aggus & Bivin	1982	156.12	RBT	H	surface area	storage ratio	H2O level fluctuatio		RP	0.55	24	13.2	28	4	O,R				y	US
Matuszek	1978	5.1	mixed	H	mean depth	TDS			RP	0.68	19	13.0	22	3	O,R,I		Avg. ann. catch (wet wt.) of the max. 15 yr. commercial catch + (the est. avg. ann. sport , domestic , and mink ranch catch for same period) was used as the MSY for mod. development. Only the best 5 of 9 regressions incl. here. See #168 for valid. info.		y	Data compiled from publ. literature; regions were not defined. Lake size ranges from 189 to 82,414 km2.
Hanson & Legett	1982	3.7	mixed	H	TP				RP	0.75	16	12.0	18	2	R,I		Variable ranges pertain to both datasets 1 AND 2 which were used to develop models (3.1, 3.2) and (3.3, 3.4) respectively. Only the most significant SR and MR models reported here. See paper for more. See models 1 (p. 1774) and 168 for validation info.	1, 168	y	lakes and ponds 0-58N, 121-122W
Matuszek	1978	5.2	LT, WF, WALL, sauger	H	mean depth	TDS			RP	0.65	18	11.7	21	3	O,R,I		Avg. ann. catch (wet wt.) of the max. 15 yr. commercial catch + (the est. avg. ann. sport , domestic , and mink ranch catch for same period) was used as the MSY for mod. development. Only the best 5 of 9 regressions incl. here. See #168 for valid. info.		y	Data compiled from publ. literature; regions were not defined. Lake size ranges from 189 to 82,414 km2.

LAKE MODELS used for RECREATIONAL/REGIONAL PLANNING, where coarse estimates of stock capacity are estimated from map-based or easily obtained information.

Author	Date	Model #	Spp.	Variables: dependent	Variables: ind. 1	Variables: ind. 2	Variables: ind. 3	Variables: ind. 4	Model Use Category	R2	DF	R2*DF	N	Terms	Man. Level	Comments on Rank	Comments on Report	Related Models	Data Avail.	Area Description
Hanson & Legett	1982	3.1	mixed	H	macro - benthos	mean depth			RP	0.48	24	11.5	26	2	R,I		Var1/ Var2 in mod. Var. ranges pertain to both datasets 3 AND 4 which were used to dev. # (3.5, 3.6) and (3.7) respectively. Only most sign. SR and MR models reported here. See paper for more. See models 1 (p. 1774) and 168 for validation info.	1, 168	n- macro - benthos	North Temperate lakes, 42-82N, 17-117E
Adams & Olver	1977	35.3	mixed, WALL	H	MEI				RP	0.70	16	11.2	18	2	O,R		RYI (tot. yield/theor. yield) was substituted for CPUE (not avail.). RYI is assumed to represent effort and vulnerability to catch. 1 model was developed for each of 4 RYI classes. This model is for lakes with Moderate Fishing Pressure, RYI 30-49%.		Y	Relatively homogenous set of 70 lakes in N.W. Ontario (80-95W, 48-54N) subject to a wide range of fishing pressure.
Hubert & Guenther	1992	204	T	SC	TDS	non-sal abundance			RP	0.56	19	10.6	22	3	I		Effects of non-salmonids and TDS on trout abundance.		Y	reservoirs <= 890 ha. full pool, elev. 1253 3166 m
McConnell	1977	135	RBT, mixed	H	gross photosyn				RP	0.93	11	10.3	13	2	O,R,I		Gross photosynthesis from all benthic and planktonic plants. See limitations on pp. 422-423.		n- gross photosyn	small ponds and pools having a high fishing pressure
Aggus & Bivin	1982	156.8	KOK	H	frost free days	storage ratio			RP	0.64	16	10.2	19	3	O,R				Y	US
Carlander	1977	2.4	Wall	H	TDS	area			RP	0.55	18	9.9	21	3	O,R		N may actually be 17; text is unclear so I have used entire data set. Data are averaged over several years.		Y	Data compiled from literature; regions include Canada and USA north of Mass.
Schlesinger & Reiger	1983	17.7	Wall	H	MEI				RP	0.69	14	9.7	18	2	O,R				Y	Intensively fished temperate and subarctic lakes in N.A.
Oglesby	1977	6.2	mixed	H	productivity				RP	0.74	13	9.6	15	2	R		Yield expressed as dry wt./m2 of lake surface or as carbon/m2 of lake surface. Variable ranges reported for entire set of 48 lakes only. It may be possible to extract subset ranges if effort is warranted. Model applicable to lakes > 10km2 only.		n- productivity	Data compiled from publ. literature; regions were not defined. Lake location ranged from 10 - 59N.
Rawson	1952	7	mixed	P	depth				RP	0.92	10	9.2	13	3	O,R		N may be only 10; see p. 519.		Y	large, extensively fished lakes
Jenkins & Morais	1971	34.1	mixed, RBT	H	growing season				RP	0.09	99	8.9	103	4	O,R,I				Y	reservoirs throughout US
Downing, Plante, & Lalonde	1990	42.2	mixed	P	TP				RP	0.79	11	8.7	14	3	R		Data collected from various published sources, so some details not included in paper (i.e. how many seasons/years the data span). PAGE MISSING From Aqu. Res. copy. I relied on their reporting of equations.		Y	wide range of geographic areas and trophic status
Schlesinger & Reiger	1983	17.6	NP	H	MEI				RP	0.61	14	8.5	16	2	O,R				Y	Intensively fished temperate and subarctic lakes in N.A.

LAKE MODELS used for RECREATIONAL/REGIONAL PLANNING, where coarse estimates of stock capacity are estimated from map-based or easily obtained information.

Author	Date	Model #	Spp.	Variables					Model Use Category	R2	DF	R2*DF	N	Terms	Man. Level	Comments on Rank	Comments on Report	Related Models	Data Avail.	Area Description
				dependent	ind. 1	ind. 2	ind. 3	ind. 4												
Hanson & Legett	1982	3.1	mixed	H	macro - benthos	mean depth			RP	0.48	24	11.5	26	2	R,I		Var1/ Var2 in mod. Var. ranges pertain to both datasets 3 AND 4 which were used to dev. # (3.5, 3.6) and (3.7) respectively. Only most sign. SR and MR models reported here. See paper for more. See models 1 (p. 1774) and 168 for validation info.	1, 168	n- macro - benthos	North Temperate lakes, 42-62N, 17-117E
Adams & Olver	1977	35.3	mixed, WALL	H	MEI				RP	0.70	18	11.2	18	2	O,R		RYI (tot. yield/theor. yield) was substituted for CPUE (not avail.). RYI is assumed to represent effort and vulnerability to catch. 1 model was developed for each of 4 RYI classes. This model is for lakes with Moderate Fishing Pressure, RYI 30-49%.		Y	Relatively homogenous set of 70 lakes in N.W. Ontario (80-95W, 48-54N) subject to a wide range of fishing pressure.
Hubert & Guenther	1992	204	T	SC	TDS	non-sal abundance			RP	0.56	19	10.6	22	3	I		Effects of non-salmonids and TDS on trout abundance.		Y	reservoirs < = 890 ha. full pool , elev. 1253 3166 m
McConnell	1977	135	RBT, mixed	H	gross photosyn				RP	0.93	11	10.3	13	2	O,R,I		Gross photosynthesis from all benthic and planktonic plants. See limitations on pp. 422-423.		n- gross photosyn	small ponds and pools having a high fishing pressure
Aggus & Bivin	1982	156.8	KOK	H	frost free days	storage ratio			RP	0.64	16	10.2	19	3	O,R				Y	US
Carl&er	1977	2.4	Wall	H	TDS	area			RP	0.55	18	9.9	21	3	O,R		N may actually be 17; text is unclear so I have used entire data set. Data are averaged over several years.		Y	Data compiled from literature; regions include Canada and USA north of Mass.
Schlesinger & Reiger	1983	17.7	Wall	H	MEI				RP	0.69	14	9.7	16	2	O,R				Y	Intensively fished temperate and subarctic lakes in N.A.
Oglesby	1977	6.2	mixed	H	productivity				RP	0.74	13	9.6	15	2	R		Yield expressed as dry wt./m2 of lake surface or as carbon/m2 of lake surface. Variable ranges reported for entire set of 48 lakes only. It may be possible to extract subset ranges if effort is warranted. Model applicable to lakes > 10km2 only.		n- productivity	Data compiled from publ. literature; regions were not defined. Lake location ranged from 10 - 59N.
Rawson	1952	7	mixed	P	depth				RP	0.92	10	9.2	13	3	O,R		N may be only 10; see p. 519.		Y	large, extensively fished lakes
Jenkins & Morais	1971	34.1	mixed, RBT	H	growing season				RP	0.09	99	8.9	103	4	O,R,I				Y	reservoirs throughout US
Downing, Plante, & Lalonde	1990	42.2	mixed	P	TP				RP	0.79	11	8.7	14	3	R		Data collected from various published sources, so some details not included in paper (i.e. how many seasons/years the data span). PAGE MISSING From Aqu. Res. copy. I relied on their reporting of equations.		Y	wide range of geographic areas and trophic status
Schlesinger & Reiger	1983	17.6	NP	H	MEI				RP	0.61	14	8.5	16	2	O,R				Y	Intensively fished temperate and subarctic lakes in N.A.

LAKE MODELS used for **RECREATIONAL/REGIONAL PLANNING, where coarse estimates of stock capacity are estimated from map-based or easily obtained information.**

Author	Date	Model #	Spp.	Variables: depend-ent	Ind. 1	Ind. 2	Ind. 3	Ind. 4	Model Use Category	R2	DF	R2*DF	N	Terms	Man. Level	Comments on Rank	Comments on Report	Related Models	Data Avail.	Area Description
Oglesby	1977	6.5	mixed	H	MEI				RP	0.70	12	8.4	14	2	R		Yield expressed as dry wt./m2 of lake surface or as carbon/m2 of lake surface. Variable ranges reported for entire set of 46 lakes only. It may be possible to extract subset ranges if effort is warranted. Model developed for ponds and reservoirs.		y	Data compiled from publ. literature; regions were not defined. Lake location ranged from 10 - 59N.
Oglesby	1977	6.4	mixed	H	MEI				RP	0.59	13	7.7	15	2	R		Yield expressed as dry wt./m2 of lake surface or as carbon/m2 of lake surface. Variable ranges reported for entire set of 46 lakes only. It may be possible to extract subset ranges if effort is warranted. Model applicable to lakes > 10km2 only.		y	Data compiled from publ. literature; regions were not defined. Lake location ranged from 10 - 59N.
Schlesinger & Reiger	1983	17.2	NP	H	long-term mean annual air temp				RP	0.66	11	7.5	16	5	O,R				y	Intensively fished temperate and subarctic lakes in N.A.
Matuszek	1978	5.5	mixed	H	bottom fauna standing crop				RP	0.83	9	7.5	11	2	O,R,I		Avg. ann. catch (wet wt.) of the max. 15 yr. commercial catch + (the est. avg. ann. sport , domestic , and mink ranch catch for same period) was used as the MSY for mod. development. Only the best 5 of 9 regressions incl. here. See #168 for valid. info.		n- bottom fauna standing crop	Data compiled from publ. literature; regions were not defined. Lake size ranges from 189 to 25,382 km2.
Schlesinger & Reiger	1983	17.5	WF	H	MEI				RP	0.49	15	7.4	17	2	O,R				y	Intensively fished temperate and subarctic lakes in N.A.
Schlesinger & Reiger	1983	17.3	Wall	H	long-term mean annual air temp				RP	0.73	10	7.3	15	5	O,R		"N" may be 16, text unclear.		y	Intensively fished temperate and subarctic lakes in N.A.
Matuszek	1978	5.4	LT, WF, Wall, sauger	H	bottom fauna standing crop				RP	0.80	9	7.2	11	2	O,R,I		Avg. ann. catch (wet wt.) of the max. 15 yr. commercial catch + (the est. avg. ann. sport , domestic , and mink ranch catch for same period) was used as the MSY for mod. development. Only the best 5 of 9 regressions incl. here. See #168 for valid. info.		n- bottom fauna standing crop	Data compiled from publ. literature; regions were not defined. Lake size ranges from 189 to 25,382 km2.
Matuszek	1978	5.3	mixed	H	bottom fauna standing crop				RP	0.80	9	7.2	11	2	O,R,I		Avg. ann. catch (wet wt.) of the max. 15 yr. commercial catch + (the est. avg. ann. sport , domestic , and mink ranch catch for same period) was used as the MSY for mod. development. Only the best 5 of 9 regressions incl. here. See #168 for valid. info.		n- bottom fauna standing crop	Data compiled from publ. literature; regions were not defined. Lake size ranges from 189 to 25,382 km2.

LAKE MODELS used for RECREATIONAL/REGIONAL PLANNING, where coarse estimates of stock capacity are estimated from map-based or easily obtained information.

Author	Date	Model #	Spp.	Variables: dependent	Variables: ind. 1	Variables: ind. 2	Variables: ind. 3	Variables: ind. 4	Model Use Category	R2	DF	R2*DF	N	Terms	Man. Level	Comments on Rank	Comments on Report	Related Models	Data Avail.	Area Description
Carl&er	1977	2.2	Wall	H	TDS				RP	0.37	19	7.1	21	2	O,R		N may actually be 17; text is unclear so I have used entire data set. Data are averaged over several years.		y	Data compiled from literature; regions include Canada and USA north of Mass.
Schlesinger & Reiger	1983	17.1	WF	H	long-term mean annual air temp				RP	0.68	10	6.8	14	4	O,R		"N" may be 17, text unclear.		y	Intensively fished temperate and subarctic lakes in N.A.
Carl&er	1977	2.3	Wall	H	area				RP	0.34	19	6.4	21	2	O,R		N may actually be 17; text is unclear so I have used entire data set. Data are averaged over several years.		y	Data compiled from literature; regions include Canada and USA north of Mass.
Carl&er	1977	2.5	Wall	H	TDS	area			RP	0.23	27	6.1	30	3	O,R		18 lakes dropped from sample of 48 because TDS not reported. Data are generally averaged over several years.		y	Data compiled from literature; regions include Canada and USA north of Mass.
Rempel & Colby	1991	138.2	mixed	H	area	vol	TDS		RP	0.08	70	5.9	74	4	O,R		Ryder 1965 data converted to SI units. Authors advise that model should be applied only to oligotrophic lakes. "P" value is my estimate from Table 2 in text.	Ryder 1965	y	sport fisheries lakes only
Aggus & Bivin	1982	156.6	CT	H	mean depth	outlet depth			RP	0.49	12	5.8	15	3	O,R				y	US
Adams & Olver	1977	35.2	mixed, WALL	H	MEI				RP	0.29	20	5.8	22	2	O,R		RYI (tot. yield/theor. yield) was substituted for CPUE (not avail.). RYI is assumed to represent effort and vulnerability to catch. 1 model was developed for each of 4 RYI classes. This model is for lakes with High Fishing Pressure, RYI 50-100%.		y	Relatively homogenous set of 70 lakes in N.W. Ontario (90-95W, 48-54N) subject to a wide range of fishing pressure.
Oglesby	1977	8.3	mixed	H	MEI				RP	0.38	15	5.7	17	2	R		Yield expressed as dry wt./m2 of lake surface or as carbon/m2 of lake surface. Variable ranges reported for entire set of 46 lakes only. It may be possible to extract subset ranges if effort is warranted. Model applicable to lakes < 25m deep only.		y	Data compiled from publ. literature; regions were not defined. Lake location ranged from 10 - 59N.
Carl&er	1977	2.1	Wall	H	latitude				RP	0.34	15	5.0	17	2	O,R		4 lakes dropped from sample of 21 (p. 1805). Data are averaged over several years.		y	Data compiled from literature; regions include Canada and USA north of Mass.
Ryder	1965	8.4	mixed	P	MEI				RP	0.55	9	4.9	11	2	O,R		Commonly cited classic. Includes commercial harvest. See models 1 (p. 1774), 58 (p.175) and 168 for validation info.	1, 168	y	N. Temp. region, alt. < 2000 ft., lake area 23-11,800 sq. mi. lightly fished
Schlesinger & Reiger	1983	17.4	LT	H	MEI				RP	0.40	12	4.8	14	2	O,R				y	Intensively fished temperate and subarctic lakes in N.A.

LAKE MODELS used for RECREATIONAL/REGIONAL PLANNING, where coarse estimates of stock capacity are estimated from map-based or easily obtained information.

Author	Date	Model #	Spp.	Variables					Model Use Category	R2	DF	R2*DF	N	Terms	Man. Level	Comments on Rank	Comments on Report	Related Models	Data Avail.	Area Description
				depend-ent	Ind. 1	Ind. 2	Ind. 3	Ind. 4												
Aggus & Bivin	1982	156.5	BKT	H	H2O level fluctuatio				RP	0.79	6	4.8	9	3	O,R				y	US
Aggus & Bivin	1982	156.7	COHO	H	H2O level fluctuatio	shore develop.			RP	0.76	6	4.5	9	3	O,R				y	US
Aggus & Bivin	1982	156.9	LT	H	outlet depth				RP	0.95	4	3.8	8	2	O,R				y	US
Rempel & Colby	1991	138.3	mixed	H	area	vol	TDS		RP	0.10	28	2.5	30	4	O,R		Ryder 1965 data converted to SI units. Authors advise that model should be applied only to oligotrophic lakes. "P" value is my estimate from Table 2 in text.	Ryder 1965	y	commercial fisheries lakes only
Rempel & Colby	1991	138.1	mixed	H	area	vol	TDS		RP	0.10	19	1.9	23	4	O,R		Ryder 1965 data converted to SI units. Authors advise that model should be applied only to oligotrophic lakes. "P" value is my estimate from Table 2 in text.	Ryder 1965	y	
Adams & Olver	1977	35.4	mixed, walleye	H	MEI				RP	0.09	17	1.6	19	2	O,R		RYI (tot. yield/theor. yield) was substituted for CPUE (not avail.). RYI is assumed to represent both and vulnerability to catch. 1 model was developed for each of 4 RYI classes. This model is for lakes with Light Fishing Pressure, RYI < 29%.		y	Relatively homogenous set of 70 lakes in N.W. Ontario (80-95W, 48-54N) subject to a wide range of fishing pressure.
Adams & Olver	1977	35.1	mixed, WALL	H	MEI				RP	0.04	9	0.4	11	2	O,R		RYI (tot. yield/theor. yield) was substituted for CPUE (not avail.). RYI is assumed to represent effort and vulnerability to catch. 1 model was developed for each of 4 RYI classes. This model is for lakes with Excessive Fishing Pressure, RYI > 100%.		y	Relatively homogenous set of 70 lakes in N.W. Ontario (80-95W, 48-54N) subject to a wide range of fishing pressure.

STREAM MODELS used for STOCK MANAGEMENT, where measured fish population status is compared to estimates of predictive capability.

Author	Date	Model #	Spp.	Variables: dependent	Variables: ind. 1	Variables: ind. 2	Variables: ind. 3	Variables: ind. 4	Model Use Category	R2	DF	R2*DF	N	Terms	Man. Level	Comments on Rank	Comments on Report	Related Models	Data Avail.	Area Description
Ptolemy et al.	1991	162.2	S	SC	fish wgt.	alkalinity	fixed non-filterable residue		SM	0.84	1589	1327.6	1592	4	I	Selected.	This model is used by BC MoE stock management unit for habitat assessments to compare measured stock abundance vs. predicted capacity. Model rejected by CJFAS and uncertain how accepted it is with the fisheries branch. Validated w/in paper.		y	diverse ecoregions of BC
Ptolemy et al.	1993	162.1	COHO	SC	fish wgt.	alkalinity			SM	0.68	61	41.5	64	3	I	Selected.	This model was derived in Ptolemy's paper, but the data source from which the model was derived was not defined clearly in the paper.		y	diverse ecoregions
Randolph & White	1984	111	RBT	SC	flow 11 days prev.				SM	0.98	38	37.2	40	2	R	Rejected: very specific analysis to predict effects of flow on fish density measured 11 days later. Not applicable to other sites.	As summarized by Fausch et al. 1988 (reported by Aqu. Res.).		y	
Rosenau & Slaney	1983	54.1	Mixed	SC	area of cover/total area	nitrate			SM	0.92	22	20.2	24	2	R		Good example of reanalysis of existing model using BC data. This approach could be applied more effectively now that there is more BC data. Model developed must substitute TDS for N as suggested by Slaney.	57	y	Data from Wyoming streams mainly, with a few BC streams
Jowett	1993	202.2	BNT	SC	water temperature	total benthic invertebrate biomass			SM	0.45	40	17.9	42	2	O,R		good comments on pro's/cons of IFIM.		n- total benthic invertebrate biomass	New Zealand
Milner et al.	1985	49.7	AS, BNT	SC	hardness	mw	md	depth variance	SM	0.53	32	16.9	38	6	I		% 46-60 cm. depth is 5th independent term. Note r2 comment for model 49.1. See paper 56 p. 175.		n- see comments	Hard and soft waters in Wales
Milner et al.	1985	49.9	AS, BNT	SC	hardness	mw	md	depth variance	SM	0.33	32	10.7	38	6	I		% 46-60 cm. depth is 5th independent term. Note r2 comment for model 49.1. See paper 56 p. 175.		n- see comments	Hard and soft waters in Wales
Milner et al.	1985	49.8	AS, BNT	SC	hardness	mw	md	depth variance	SM	0.22	32	7.0	38	6	I		% 46-60 cm. depth is 5th independent term. Note r2 comment for model 49.1. See paper 56 (p.175).		n- see comments	Hard and soft waters in Wales
Milner et al.	1985	49.5	AS, BNT	SC	altitude	hardness	md	depth variance	SM	0.80	6	6.4	16	8	I		0-15 cm. % depth, 46-60 cm. % depth, and cover index are last 3 independent variables. Note r2 comment for model 49.1. See paper 56 (p.175).		n- see comments	hardwater only (> 25 mg/l CaCO3), Wales
Hunt	1979	76	BKT	SC	msf				SM	0.99	6	5.9	8	2	R		As summarized by Fausch et al. 1988 (reported by Aqu. Res.).		y	

STREAM MODELS used for STOCK MANAGEMENT, where measured fish population status is compared to estimates of predictive capability.

Author	Date	Model #	Spp.	Variables: dependent	Variables: ind. 1	Variables: ind. 2	Variables: ind. 3	Variables: ind. 4	Model Use Category	R2	DF	R2*DF	N	Terms	Man. Level	Comments on Rank	Comments on Report	Related Models	Data Avail.	Area Description
Milner et al.	1985	49.2	AS, BNT	SC	altitude	hardness	mw		SM	0.49	12	5.9	16	4	I		Note that values for r2 for all Milner models is actually % variance accounted for = 100*(total mean square-residual mean square)/total mean square. This statistic takes into account the # of parameters in the model. See paper 58 (p.175).		y	hardwater only (> 25 mg/l CaCO3), Wales
Milner et al.	1985	49.8	AS, BNT	SC	altitude	mw	md	depth variance	SM	0.94	6	5.6	16	10	I		0-15 cm. % depth, 46-60 cm. % depth, % boulders, % cobbles, and cover index are last 5 independent variables. Note r2 comment for model 49.1. See paper 58 (p.175).		n- see comments	hardwater only (> 25 mg/l CaCO3), Wales
Enk	1977	68	BKT	SC	lub				SM	0.93	6	5.6	8	2	I		As summarized by Fausch et al. 1988 (reported by Aqu. Res.).		y	
Platts & Nelson	1989	142.2	5	SC	Thermal input (btu/m2/day)				SM	0.33	15	5.0	17	2	R		Only significant relationships between biomass and thermal input were entered. Note how separating data by region increases prediction capabilities of model.		y	Great Basin and Rocky Mountains (Idaho, Nevada, Utah)
Platts & Nelson	1989	142.1	S	SC	Thermal input (btu/m2/day)				SM	0.91	5	4.6	7	2	R		Only significant relationships between biomass and Thermal input were entered. Note how separating data by region increases prediction capabilities of model.		y	Great Basin (Nevada, Utah)
McFadden & Cooper	1962	95	mixed	SC	c				SM	0.72	3	2.2	5	2	I		As summarized by Fausch et al. 1988 (reported by Aqu. Res.).		y	

STREAM MODELS used for RECREATIONAL/REGIONAL PLANNING, where coarse estimates of stock capacity are estimated from map-based or easily obtained information.

Author	Date	Model #	Spp.	Variables: dependent	ind. 1	ind. 2	ind. 3	ind. 4	Model Use Category	R2	DF	R2*DF	N	Terms	Man. Level	Comments on Rank	Comments on Report	Related Models	Data Avail.	Area Description
Jowett	1992	202.1	BNT	SC	water temperature	ratio of mean/median flow	% lake area	% flat slope	RP	0.44	85	37.6	89	4	O,R	Selected.	good comments on pro's/cons of IFIM.		y	New Zealand
Lanka et al.	1987	83.1	BNT,RBT,BKT,CT	SC	rated reach elev.	relief ratio	drainage density	avg reach width	RP	0.51	60	30.6	65	5	O,R	Selected.	Forest streams		y	Colorado and Missouri river drainages in WYO
Lanka	1985	82.2	T	SC	rated reach elev.	relief ratio	drainage density	avg reach width	RP	0.51	60	30.6	65	5	O	Same as model 83.1	Trout in forest streams. As summarized by Fausch et al. 1988 (reported by Aqu. Res.).		y	
Lanka et al.	1987	83.3	BNT,RBT,BKT,CT	SC	rated reach elev.	basin ratio	relief ratio		RP	0.36	61	22.0	65	4	O,R	(Similar to model 83.1)	Forest streams		y	Colorado and Missouri river drainages in WYO
Marshall & Britton	1990	24.2	COHO	H	stream length (km)				RP	0.88	22	19.4	24	2	R		Lakes excluded from their analysis. SE reported is SE of slope. For power functions, ranges are transformed from log values to normal. Only models for all data are included in database, although separate models for small streams are also given in paper.		y	Pacific Northwest coastal streams, ponds, and sidechannels. Many sites on Vancouver Island
Lanka et al.	1987	83.2	BNT,RBT,BKT,CT	SC	avg reach width	width: depth ratio	gradient		RP	0.31	61	18.9	65	4	O,R	(Similar to model 83.1)	Forest streams		y	Colorado and Missouri river drainages in WYO
Lanka et al.	1987	83.4	BNT,RBT,BKT,CT	SC	rated midrange basin elev.	basin perimeter	channel slope	width: depth ratio	RP	0.64	21	13.4	26	5	O,R	(Similar to model 83.1)	Rangeland streams		y	Colorado and Missouri river drainages in WYO
Lanka	1985	82.1	T	SC	bp	g	rmbe	rwdr	RP	0.64	21	13.4	26	5	O		Trout in rangeland streams. As summarized by Fausch et al. 1988 (reported by Aqu. Res.).		y	
Chisholm & Hubert	1986	65	BKT	SC	md	mw	sg	wdr	RP	0.69	19	13.1	24	5	R		As summarized by Fausch et al. 1988 (reported by Aqu. Res.).		y	
Lanka et al.	1987	83.5	BNT,RBT,BKT,CT	SC	width: depth ratio	avg reach velocity			RP	0.57	23	13.1	26	3	O,R		Rangeland streams		y	Colorado and Missouri river drainages in WYO
Marshall & Britton	1990	24.4	COHO	SC	stream length (km)				RP	0.92	14	12.9	16	2	R		Lakes excluded from their analysis. SE reported is SE of slope. For power functions, ranges are transformed from log values to normal. Only models for all data are included in database, although separate models for small streams are also given in paper.		y	Pacific Northwest (Ak. to Or.)coastal streams, ponds, and sidechannels. Many sites on Vancouver Island
Marshall & Britton	1990	24.1	COHO	H	area (m2)				RP	0.83	15	12.4	17	2	R		Lakes excluded from their analysis. SE reported is SE of slope. For power functions, ranges are transformed from log values to normal. Only models for all data are included in database, although separate models for small streams are also given in paper.		y	Pacific Northwest (Ak. to Or.)coastal streams, ponds, and sidechannels. Many sites on Vancouver Island

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March 28, 1994

STREAM MODELS used for RECREATIONAL/REGIONAL PLANNING, where coarse estimates of stock capacity are estimated from map-based or easily obtained information.

Author	Date	Model #	Spp.	Variables: dependent	Variables: Ind. 1	Variables: Ind. 2	Variables: Ind. 3	Variables: Ind. 4	Model Use Category	R2	DF	R2*DF	N	Terms	Man. Level	Comments on Rank	Comments on Report	Related Models	Data Avail.	Area Description
Lanka et al.	1987	83.6	BNT,RBT,BKT, CT	SC	rated midrange basin elev.	basin relief			RP	0.52	23	12.0	28	3	O,R		Rangeland streams		y	Colorado and Missouri river drainages in WYO
Marshall & Britton	1990	24.3	COHO	SC	area (m2)				RP	0.91	13	11.9	15	2	R		Lakes excluded from their analysis. SE reported is SE of slope. For power functions, ranges are transformed from log values to normal. Only models for all data are included in database, although separate models for small streams are also given in paper.		y	Pacific Northwest (Ak. to Or.)(coastal streams, ponds, and sidechannels. Many sites on Vancouver Island
Ziemer	1973	127	PS	SC	dd	mbl	mbe		RP	0.34	19	6.5	21	2	O		As summarized by Fausch et al. 1988 (reported by Aqu. Res.).		y	
Burns	1971	25.1	S	SC	aa				RP	0.81	5	4.1	7	2	R		As summarized by Fausch et al. 1988 (reported by Aqu. Res.).		y	
Burns	1971	25.2	mixed	SC	aa				RP	0.80	5	4.0	7	2	R		As summarized by Fausch et al. 1988 (reported by Aqu. Res.).		y	
Burton & Wesche	1974	41	T	SC	da	fa	mbe	tel	RP	0.33	9	3.0	11	2	O		As summarized by Fausch et al. 1988 (reported by Aqu. Res.).		y	
Milner et al.	1985	49.1	AS, BNT	SC	altitude				RP	0.16	17	2.8	19	2	I		Note that values for r2 for all Milner models is actually % variance accounted for = 100*(total mean square-residual mean square)/total mean square. This statistic takes into account the # of parameters in the model. See paper [illegible]		y	hardwater only (> 25 mg/l CaCO3), Wales

STREAM MODELS used for HABITAT RESTORATION AND IMPROVEMENT, where predictions of benefits from managed changes in habitat complexity (e.g., stream fertilization) are required.

Author	Date	Model #	Spp.	Variables: dependent	Variables: ind. 1	Variables: ind. 2	Variables: ind. 3	Variables: ind. 4	Model Use Category	R2	DF	R2*DF	N	Terms	Man. Level	Comments on Rank	Comments on Report	Related Models	Data Avail.	Area Description
Platts (see comments)	1979	109	RBT	SC	*not specified - see comments*				HR	0.35	270	34.5	291	21	I	Rejected: seems over-parameterized which has serious implications in terms of its predictive abilities in systems other than where it was developed.	As summ. by Fausch et al. 1988 (rev. w/ Platts 1974). Uses 20 variables (3 types: drainage basin; channel morphometry & flow; and habitat structure/ bio/ chem/ phys), but elev, stream width, and pool rating explained the most variation.		u	
Fraley & Graham	1981	69	CT, Bull	SC	overhead cover	instream cover	stream cover	substrate	HR	0.64	130	83.2	134	4	I	Selected.	Fausch reports R2 as 0.41. Can't find this in text. Model would still rate in top ten stream capability models if R2 reduced.			Flathead River drainage
Barber et al.	1981a	61.1	COHO	SC	area w/ substrate dia. 8-256 mm.	area of overhang rip. veg.	days elapsed from June 1		HR	0.76	72	54.7	76	4	I	(Selected) Note: same as model 105.1.	As summarized by Fausch et al. 1988 (reported by Aqu. Res.).		y	
Orwood & Barber	1982	105.1	COHO	SC	area w/ substrate dia. 8-256 mm.	area of overhang rip. veg.	days elapsed from June 1		HR	0.76	72	54.7	76	4		(Selected) Note: same as model 61.1	Coho age class 0. Same as model 61.1.		y	12 first to third order streams in southeastern Alaska.
Orwood & Barber	1982	105.3	DV	SC	surface area	area w/forest debris in riffles			HR	0.49	73	35.8	76	3		Selected.			y	12 first to third order streams in southeastern Alaska.
Binns & Eiserman	1979	38	T	SC	*not specified - see comments*				HR	0.97	30	29.1	38	8	I		As summ. by Fausch et al. 1988 (w/ #37), this HQI model, was validated by B&I (8 sites) and did well. Model used 5 variables (3 types: drainage basin; channel morphometry & flow; and habitat structure/ bio/ chem/ phys) . See paper 58 p.175.	37	u	
Nickelson et al.	1979	101.1	CT	SC	*not specified - see comments*				HR	0.91	27	24.6	29	2	I		As summarized by Fausch et al. 1988, this model uses one variable, "total cover", by summing the freq. of suitable depths, instream cover, overhanging cover, surf. turb., and velocity refuge. See Fausch for more details.		u	
Nickelson et al.	1979	101.2	SHT	SC	*not specified - see comments*				HR	0.79	21	16.6	23	2	I		As summarized by Fausch et al. 1988, this model is similar to model 101.1 (different fish spp.).		u	
Ward & Slaney	1980	122	SHT	SC	rr.30				HR	0.69	13	9.0	15	2	I		As summarized by Fausch et al. 1988 (reported by Aqu. Res.).		y	
Klamt	1976	79	S	SC	id	pbl			HR	0.38	12	4.3	15	3	I		As summarized by Fausch et al. 1988 (reported by Aqu. Res.).		n- id	
Wesche	1980	123.2	BNT	SC	arbv	lub	prbv	pub	HR	0.60	6	3.6	8	2	I		As summarized by Fausch et al. 1988 (reported by Aqu. Res.).		n- prbv, pub	

STREAM MODELS used for HABITAT IMPACTS AND MITIGATION, where predictions of impacts of logging and flow reductions are required.

Author	Date	Model #	Spp.	Variables: dependent	ind. 1	ind. 2	ind. 3	ind. 4	Model Use Category	R2	DF	R2*DF	N	Terms	Man. Level	Comments on Rank	Comments on Report	Related Models	Data Avail.	Area Description
Sekulich	1980	114	CHIN	SC	*not specified - see comments*				III	0.89	76	67.6	80	4	I	Selected: though the egg deposition input requirements of this model would limit its potential application.	As summarized by Fausch et al. 1988, variables included # eggs deposited, channel morphometry and flow, temperature, and biological variables.		u	
Hershberger & Bhattacha.	1981	88	T	SC	ac	sov,1	sov1-2	rv	HI	0.66	93	61.4	100	3	I	Rejected: 18 variables describing trout cover were reduced to 6 factors using factor analysis. Regressions done using 6 factors as independent variables. Relationship not transferrable. Very detailed cover estimates which are subjectively measured.	% of pool area as brush cover and % of area as instream bank veg. are the remaining 2 of 6 variables used in model. As summarized by Fausch et al. 1988 (reported by Aqu. Res.).		y	
Jowett	1984	202.3	BNT	SC	water temperature	% WUA derived from depth, velocity, and substrate	instream trout cover grade	gradient	HI	0.82	50	41.0	59	9	I	Rejected: used WUA.	good comments on pro's/cons of IFIM. 4 other independent variables are % sand substrate, % WUA for food production, % lake area, elevation, % development of pasture crop		n- see comments	New Zealand
Barber et al.	1981a	61.2	CT	SC	chan. width bnkfull ft.	bank stability			HI	0.56	73	40.9	76	3	I	Selected.	As summarized by Fausch et al. 1988 (reported by Aqu. Res.).		y	
Barber et al.	1981a	61.3	DV	SC	pool width	riffle width			HI	0.54	73	39.4	76	3	I	Selected	As summarized by Fausch et al. 1988 (reported by Aqu. Res.).		y	
Barber et al.	1981a	61.4	COHO	SC	gradient	area where D < 0.5 m, v <0.3 m/s	area of overhang rip. veg.	area of undercut banks	HI	0.49	71	34.8	76	5	I	(Selected) Note: same as model 105.2	As summarized by Fausch et al. 1988 (reported by Aqu. Res.).		y	
Dewood & Barber	1982	105.2	COHO	SC	gradient	area where D < 0.5 m, v <0.3 m/s	area of overhang rip. veg.	area of undercut banks	HI	0.49	71	34.8	76	5		(Selected) Note:same as model 61.4	Coho age classes 1+. Same as model 61.4.		y	12 first to third order streams in southeastern Alaska.
Dewood & Barber	1982	105.4	T	SC	area w/forest debris in riffles	area where D > 0.5 m, V >0.3 m/s	area w/forest debris in pools	area of overhang rip. veg. in riffles	HI	0.43	71	30.5	76	5		(Selected) Note: similar to 61.4			y	12 first to third order streams in southeastern Alaska.
Marshall	1985	93	BNT	SC	*not specified - see comments*				HI	0.86	19	16.3	24	5	I		As summarized by Fausch et al. 1988, this model predicts fall biomass of fingerling BT from 4 variables (3 types: channel morphometry; cover; and substrate)		u	

STREAM MODELS used for HABITAT IMPACTS AND MITIGATION, where predictions of impacts of logging and flow reductions are required.

Author	Date	Model #	Spp.	Variables: dependent	Variables: ind. 1	Variables: ind. 2	Variables: ind. 3	Variables: ind. 4	Model Use Category	R2	DF	R2*DF	N	Terms	Man. Level	Comments on Rank	Comments on Report	Related Models	Data Avail.	Area Description
Stewart	1970	117.1	RBT	SC	*not specified - see comments*				HI	0.43	37	15.9	41	4	I		As summarized by Fausch et al. 1988, this model predicts RBT standing crop from 3 variables (2 types: mean depth and cover).		u	
Jowett	1995	202.4	BNT	SC	water temperature	total benthic invertebrate biomass	% WUA derived from depth, velocity, and substrate		HI	0.64	24	15.5	27	3	I		good comments on pro's/cons of IFIM.		n- total benthic invertebrate biomass	New Zealand
Leathe & Enk	1985	86.2	CT	SC	*not specified - see comments*				HI	0.46	31	14.3	35	3	I		As summarized by Fausch et al. 1988, this model which uses 3 independent variables (incl. instream cover and some type of substrate-condition score) was developed to explore the effects of small hydropower projects.		u	
Stewart	1970	117.2	BKT	SC	*not specified - see comments*				HI	0.37	38	14.1	41	3	I		As summarized by Fausch et al. 1988, this model predicts BKT standing crop from 2 variables (2 types: mean depth and cover).		u	
Konopacky	1984	80	CHIN	SC	*not specified - see comments*				HI	0.82	17	13.9	21	4	I		As summarized by Fausch et al. 1988 this model uses 3 variables of channel morphometry and stream substrate.		u	
Stalnaker	1979	118	BNT	SC	wua				HI	0.81	17	13.8	19	2	I		As summarized by Fausch et al. 1988 (NOT reported by Aqu. Res.). In order to apply model, need to have depth, velocity, and substrate data, and SI for spp. See paper 56 p. 175.		y	
Leathe & Enk	1985	86.1	BULL T	SC	*not specified - see comments*				HI	0.81	15	12.2	21	3	I		As summarized by Fausch et al. 1988, this model which uses 5 independent variables (incl. instream cover and some type of substrate-condition score) was developed to explore the effects of small hydropower projects.		u	
Nickelson et al.	1979	101.4	SHT	SC	wua				HI	0.52	21	10.9	23	2	I		As summarized by Fausch et al. 1988 (NOT reported by Aqu. Res.). In order to apply model, need to have depth, velocity, and substrate data, and SI for spp.		y	
Lewis	1969	89	T	SC	rr				HI	0.68	16	10.9	19	3	I		As summarized by Fausch et al. 1988 (reported by Aqu. Res.).		u	
Kraft	1972	81	BKT	SC	f	mpd	mpv	pa	HI	0.83	13	10.8	20	7	R		Max pool vel. and area of all cover are the remaining 2 of 6 variables used in model. As summarized by Fausch et al. 1988 (reported by Aqu. Res.).		y	

STREAM MODELS used for HABITAT IMPACTS AND MITIGATION, where predictions of impacts of logging and flow reductions are required.

Author	Date	Model #	Spp.	Variables					Model Use Category	R2	DF	R2*DF	N	Terms	Man. Level	Comments on Rank	Comments on Report	Related Models	Data Avail.	Area Description
				dependent	Ind. 1	Ind. 2	Ind. 3	Ind. 4												
Eifert & Wesche	1982	87	T	SC	erbr				HI	0.71	15	10.7	17	2	I		As summarized by Fausch et al. 1988 (reported by Aqu. Res.).		y	
Nickelson et al.	1979	101.3	COHO	SC	rr				HI	0.94	10	9.4	12	2	I		As summarized by Fausch et al. 1988 (reported by Aqu. Res.).		u	
Rinne	1978	112	T	SC	*not specified - see comments*				HI	0.52	17	8.8	24	7	I		As summarized by Fausch et al. 1988, this model uses 6 variables of describing channel morphometry and cover.		u	
Leathe & Enk	1985	86.3	BKT	SC	*not specified - see comments*				HI	0.28	31	8.7	33	3	I		As summarized by Fausch et al. 1988, this model which uses 1 independent variables (derived from instream cover and/or some type of substrate-condition score) was developed to explore the effects of small hydropower projects.		u	
Stowell et al.	1983	118.5	SHT	SC	pe	pe2			HI	0.87	9	7.8	12	3	I		As summarized by Fausch et al. 1988 (reported by Aqu. Res.).		y	
Nickelson et al.	1979	101.5	CT	SC	wua				HI	0.27	27	7.3	29	2	I		As summarized by Fausch et al. 1988 (NOT reported by Aqu. Res.). In order to apply model, need to have depth, velocity, and substrate data, and SI for spp.		y	
Milner et al.	1985	49.3	AS, BNT	SC	hardness	da	f		HI	0.54	12	6.5	16	4	I		Note that values for r2 for all Milner models is actually % variance accounted for = 100*(total mean square-residual mean square)/total mean square. This statistic takes into account the # of parameters in the model. See paper 56 (p.175).		y	hardwater only (> 25 mg/l CaCO3), Wales
Wesche	1980	123.3	BNT	SC	wua				HI	0.82	7	5.7	9	2	I		As summarized by Fausch et al. 1988 (NOT reported by Aqu. Res.). In order to apply model, need to have depth, velocity, and substrate data, and SI for spp.		y	
Stowell et al.	1983	118.4	SHT	SC	pe				HI	0.90	6	5.4	8	2	I		As summarized by Fausch et al. 1988 (reported by Aqu. Res.).		y	
Scarnecchia	1983	113.1	T	SC	*not specified - see comments*				HI	0.84	6	5.0	11	5	I		As summarized by Fausch et al. 1988, this model uses 4 habitat variables (such as elevation, width-to-depth ratio, substrate diversity, nitrate, and sulfate).		u	
Stowell et al.	1983	118.1	CHI	SC	pe				HI	0.99	5	5.0	7	2	I		As summarized by Fausch et al. 1988 (reported by Aqu. Res.).		y	
White et al.	1976	125.1	T	SC	msf	mwf	xspf		HI	0.95	5	4.8	9	4	R		As summarized by Fausch et al. 1988 (reported by Aqu. Res.).		y	
White et al.	1976	125.2	T	SC	msf	pec			HI	0.94	5	4.7	9	4	R		As summarized by Fausch et al. 1988 (reported by Aqu. Res.).		y	
Stowell et al.	1983	118.3	CHI	SC	pe	pe2			HI	0.92	5	4.6	8	3	I		As summarized by Fausch et al. 1988 (reported by Aqu. Res.).		y	
Scarnecchia	1983	113.2	T	SC	*not specified - see comments*				HI	0.76	6	4.6	11	5	I		As summarized by Fausch et al. 1988, this model uses 4 habitat variables (such		u	
Hawkins et al.	1983	74.1	S	SC	pe,1				HI	0.72	6	4.3	8	2	I		As summarized by Fausch et al. 1988 (reported by Aqu. Res.).		y	

APPENDIX B ABSTRACTS OF PROCESS

STREAM MODELS used for HABITAT IMPACTS AND MITIGATION, where predictions of impacts of logging and flow reductions are required.

Author	Date	Model #	Spp.	Variables: dependent	ind. 1	ind. 2	ind. 3	ind. 4	Model Use Category	R2	DF	R2*DF	N	Terms	Man. Level	Comments on Rank	Comments on Report	Related Models	Data Avail.	Area Description
Nehring & &erson	1983	96.1	BNT	SC	wua				HI	0.96	3	2.9	5	2	I		As summarized by Fausch et al. 1988 (NOT reported by Aqu. Res.). Reviewed together w/#99 as model 98, and reviewed w/#96-99 as model 96. In order to apply model, need to have depth, velocity, and substrate data, and SI for spp.		y	
Nehring & &erson	1983	96.2	RBT	SC	wua				HI	0.92	3	2.8	5	2	I		As summarized by Fausch et al. 1988 (NOT reported by Aqu. Res.). Reviewed together w/#99 as model 98, and reviewed w/#96-99 as model 96. In order to apply model, need to have depth, velocity, and substrate data, and SI for spp.		y	
Nehring & &erson	1981	96.3	BNT	SC	dwf3p				HI	0.88	3	2.6	5	2	R		As summarized by Fausch et al. 1988 (reported by Aqu. Res.).		y	
Nelson	1980	100	T	SC	nsf				HI	1.00	2	2.0	4	2	R		As summarized by Fausch et al. 1988 (reported by Aqu. Res.).		y	
Stowell et al.	1983	118.2	SHT	SC	pe2	pe3			HI	0.99	2	2.0	5	3	I		As summarized by Fausch et al. 1988 (reported by Aqu. Res.).		y	
Nehring & &erson	1981	96.2	BNT	SC	xsf				HI	0.98	2	2.0	4	2	R		As summarized by Fausch et al. 1988 (reported by Aqu. Res.).		y	
Nehring & &erson	1981	96.1	RBT	SC	xsf1p				HI	0.98	2	2.0	4	2	R		As summarized by Fausch et al. 1988 (reported by Aqu. Res.).		y	
Gordon & MacCrimmon	1982	70	S	SC	pic				HI	0.98	2	2.0	4	2	I		As summarized by Fausch et al. 1988 (reported by Aqu. Res.).		n -pic	
Wesche	1980	123.1	BNT	SC	adw	arbv	lub	prbv	HI	0.95	2	1.9	4	2	I		Preference factor for undercut bank is the 5th variable used in model. As summarized by Fausch et al. 1988 (reported by Aqu. Res.).		n- adw, prbv, pub	
Milner et al.	1985	49.4	AS, BNT	SC	mw	md	depth variance	% 0-15 cm depth	HI	0.13	10	1.3	16	6	I		Cover index is 5th independent variable. Note r2 comment for model 49.1. See paper 58 (p.175).		n - % 0-15 cm depth	hardwater only (> 25 mg/l CaCO3), Wales

1.1 FISHREGS
Espegren *et al.* 1990

FISHREGS is a computer model designed to simulate the effects of various angling regulations (size and slot limits) on fish populations. The model accounts for recruitment, fish growth rate, a number of mortality factors, and population size and age structure. Parameterization of the model can range from very simplistic to extremely complex depending upon the amount of information known about the population under consideration. The model uses a saturating function to predict egg numbers at high spawner densities. FISHREGS requires age-specific, average fish length and size ranges. Within-cohort growth distributions are simulated using a beta distribution and can assume various shapes. Natural and fishing mortality parameters are estimated by combining life table and creel survey data. Life table data provides estimates of within-cohort total annual mortality. Creel survey data provides estimates of within-cohort mortality under the angling regulation currently in effect. The model is calibrated by revising the mortality rates to match available creel survey and life table data collected in the field. The model has been applied to rainbow trout and Brown trout in Colorado streams.

1.2 Small Lakes Integrated Management Model (SLIM)
Korman *et al.* 1993b

The motivation for creating SLIM is to provide a tool for regional management biologists to assess alternative management actions such as harvesting and stocking policies at both site (single-lake) and regional (multi-lake) scales. SLIM contains 3 basic elements:

1) a dynamic age-structured salmonid population model which simulates the response of wild and hatchery populations to management actions influencing density and age-specific mortality;

2) a linkage to a database of physical, limnological and biological information for over 3,000 lakes in BC(Lakes Database) which is used to provide input information required by the population model for lake-specific simulations; and

3) a Graphical User Interface (GUI) which allows SLIM user's to implement different management actions, alter assumptions and / or structural relationships within the population model, access the database, and view the results of model simulations.

Version 2.0 of SLIM has recently been completed and incorporates refinements suggested by user's following the first release of the model. Version 2.0 is currently being used and evaluated by a 'test' group of regional managers (Parkinson, pers. comm.).

1.3 Large Lakes Kokanee Model (LLKM)
Korman *et al.* 1993a
Korman and Parkinson 1993

LLKM is a dynamic age-structured fisheries model designed to examine a variety of management actions and problems common to kokanee fisheries in the Pacific Northwest. The model can be used to examine the effects of harvest regulations, predator introductions, spawning channels, sockeye enhancement, and lake fertilization on kokanee populations. The five major components of the kokanee model simulate:

- kokanee population dynamics;
- competition between kokanee and whitefish or sockeye;
- predator population dynamics;
- predation on kokanee and competitors; and
- fishing mortality on kokanee and predators.

LLKM tracks the growth and survival of stream-, shore-, and channel-spawning populations and hatchery fish on and independent basis. Changes in growth and survival are calculated on an annual timestep. Intra- and interspecific competition are modelled based on a semi-empirical bioenergetic approach which is based on food conversion efficiencies of each size class. The growth response to changes in kokanee production and lake productivity is determined based on an empirical function similar to the one derived by Rieman and Meyers (1992).

LLKM uses three alternative methods for simulating predator dynamics. Predator size is not dynamic, but densities are either: 1) controlled by a Ricker-type response in juvenile survival to adult biomass (cannibalism); 2) controlled by a Beverton-Holt-type response in yearling production to egg deposition (rearing-habitat limitation); or are 3) held constant for the duration of a simulation.

The consumption of kokanee is dependent on the biomass gain between predator size classes and their food conversion efficiencies. Faster growing and/or longer-lived predators will, over their lifetime, eat more kokanee than slower growing and/or shorter-lived ones. The predators maximum kokanee diet is reduced based on a logistic function of total kokanee density; the greater the density, the closer the predator will come to achieving its maximum requirement.

The raw output of the model is a time series plot of various indicators such as population densities, length at age, CPUE, etc. The model has been applied to (Korman and Parkinson, 1993): 1) assess the impacts of a spawning channel on natural populations of kokanee in Kootenay Lake; 2) the impact of alternative Gerrard trout yearling production on kokanee in Kootenay Lake; 3) assess the effects of kokanee density and fishing effort on fishing quality in Okanagan Lake; and to 4) to quantify verbal models of the processes that are felt to be important vis-a-vis increasing kokanee production in Williston Lake.

1.4 MANSIM: A fairly generic computer model for simulation management actions on single fish populations (Version 3.0)
Korver 1992b

MANSIM is a generalized computer simulation model for assessing the potential effects of a number of management actions on fish populations. The model has been used as a basis for constructing a harvest management model for walleye in eastern Lake Ontario. Features of the model include: assessment of effects of changes in nominal fishing effort, fishing seasons, bag and size-limits, fishable and protected slots; a monthly time step for calculating effects of growth, fishing and natural mortality; annual data summaries; separate density-dependent growth terms for age-1 recruits and the rest of the population; and a generalized stock-recruitment function with a random error-term; and separate evaluation of "trophy"-sized fish.

1.5 Lake Trout Management Support System (LTMSS) Korver 1992a

LTMSS is a customized, "home-grown" software package designed to assist fisheries biologists and managers in making decisions regrading the management and assessment of lake trout lakes in Ontario. It contains several important features specifically included to make it useful to Ontario Ministry of Natural Resources field staff throughout the province:

1) **It is general**. LTMSS is designed to work for all lake trout lakes in Ontario between 50 and 25,000 ha in surface area.

2) **It is Flexible**. You can run LTMSS even if all you know about the lake is its surface area and a "ball-park" estimate of summer and winter effort. Whenever you have more information such as growth data, maturity, stocking rates, etc., it can accommodate this information. Obviously the more information you have, the more accurate the output of LTMSS is likely to be.

3) **It is Comprehensive**. The primary purpose of LTMSS is to allow its users to ask "what if" style management questions. A wide range of management options can be examined, including effort controls, size-based regulations, and stocking.

LTMSS is a product of the Lake Trout Synthesis. The model consists of a generalized lake trout population (simulation) model which separately accounts for both stocked and naturally produced populations, embedded within a menu-based framework that allows user control of both inputs to and outputs from the model.

1.6 Stochastic Life-Cycle Model (SLCM) Lee and Hyman 1992

SLCM is a stochastic process model which simulates the life cycle of anadromous salmonids and is designed to mimic the basic mechanisms regulating populations of Pacific salmon, while capturing some of the intra-annual and interannual variation inherent in these populations. The model was designed for population viability assessments combining advanced modelling techniques with concepts from the field of conservation biology.

While the basic structure of SLCM is similar to other life-cycle models, it differs in several ways. First, SLCM incorporates stochastic or probabilistic processes at each step in the life cycle. The binomial distribution is used extensively to introduce demographic stochasticity in survival; the beta distribution is used to introduce environmental stochasticity. Because of its stochastic nature, the model's predictions must be expressed in probabilistic terms. Multiple games are run using a Monte Carlo approach to generate probability distributions for future outcomes. Second, the model is designed to use inputs from more detailed models for specific life stages, in combination with a minimum number of empirically based parameters. SLCM users can choose among alternative models for the more contentious life stages, such as juvenile migration and adult harvest, incorporating the results of their preferred models. An ancillary calibration model has been developed that allows the SLCM to be fitted to a historical time trace of population estimates, constraining expectations of survival and their variances to historical levels.

The model also allows considerable flexibility in describing the dynamics of juvenile production. Users can choose among three density-dependent relationships to describe egg-to-smolt survival, including the Beverton-Holt, the Ricker, and a logistic response function, or use empirically based conditional probabilities. A variety of scenarios involving hatchery and natural production are possible, ranging from natural production only, to a combination of hatchery and natural production involving supplementation of adults, fry, or smolts. Allocation of naturally produced and hatchery-produced adults among terminal harvest and hatchery and natural spawning follows a set of adjustable rules that affords priority to natural escapement and hatchery broodstock needs.

1.7 Skeena River Steelhead Trout Capacity Model Tautz *et al.* 1992

Estimation of steelhead carrying capacity for the Skeena River involved the construction of a number of submodels organized into three general areas: distribution, fish use, and fish productivity. Stream order, measured from 1:50,000 scale maps in conjunction with a climatic index based on annual water yield and juvenile surveys were used to provide estimates of the lengths of systems containing steelhead. Total stream area at low summer flow for each of these reaches was estimated based on a relationship between stream width at mean annual discharge. In the next step, total area was adjusted downward to estimate useable area based on habitat suitability where useable area is predicted as a function of mean annual discharge and low flow stage (derived from a large B.C. data set).

Having obtained estimates of useable area, carrying capacity was estimated by calculating the number of smolts expected per unit of habitat. Two approaches based on smolt production values derived from the Keogh River were used:

1) a linear model which extrapolates the number of Keogh adults (or smolts) produced per kilometer of stream length to the Skeena systems; or
2) a similar model which extrapolates the number of adults or smolts produced per m^2 of total habitat and usable habitat to obtain estimates for the Skeena;

Two methods were used to adjust estimates from 1) or 2) to account for productivity differences between the Keogh and Skeena Rivers. A regression model of steelhead standing crop *vs.* total alkalinity was used to develop a calibration factor which accounted for different nutrient levels between the two systems. An alternate method based on a regression between required territory size and fish length was used to estimate the space required per fish. This regression, in conjunction with life history information was used to derive an adjustment factor for smolt production estimates/ unit area or length.

1.8 Modelling the response of native steelhead to hatchery supplementation programs in an Idaho river
Byrne *et al.* 1992

A life history model was used to predict the response of native steelhead Oncorhynchus mykiss in the Lochsa River, Idaho, (Idaho), to long-term supplementation with hatchery fry and smolts. The four key factors affecting the response of the native fish to a stocking program were (1) the number of native spawners, (2) the number of stocked fish, (3) the number and fitness of progeny from stocked fish, and (4) the amount of mating between hatchery and native fish. Long-term stocking of fry or smolts led to the extinction of native fish in some scenarios. The model can be used to help assess the risks and benefits of proposed stocking programs.

1.9 Dynamics of a northern squawfish population and the potential to reduce predation on juvenile salmonids in a Columbia River reservoir (Oregon, USA)
Rieman and Beamesderfer 1990

Northern squawfish Ptychocheilus oregonensis prey on salmonid smolts (Oncorhynchus spp.) in Columbia River reservoirs. We used simulation models to determine the potential influence of exploitation of northern squawfish on that predation. We also used correlation analysis to examine factors that may influence predation through predator recruitment. We based our simulations on estimates of mortality, relative year-class strength, and growth made from a 4-year study of resident fish predators in John Day Reservoir. Simulated predation declined with exploitation of fish longer than 275 mm (fork length) such that sustained exploitation of 10-20% annually reduced predation by 50% or more. The magnitude of change was related to the type of reproductive compensation. Recruitment was not obviously related to any environmental variable we examined, although year-class strength was negatively correlated with concurrent year-class strength of walleye Stizostedion vitreum. We believe that limited, but sustained, exploitation of northern squawfish provides an alternative to more radical control measures. We are uncertain about the potential recovery rate of exploited northern squawfish populations, however, and there is some risk that unsustained exploitation could aggravate predation. Any control program should evaluate density-dependent responses of predators.

1.10 Stocking strategies for fingerling walleyes: An individual-based model approach
Madenjian *et al.* 1991

The success of any program for stocking walleye (Stizostedion vitreum vitreum) fingerlings is strongly dependent on growth of the stocked fish during the summer and early fall months immediately following their release into lakes, reservoirs, or rivers. An individual-based model (IBM) was developed to describe growth of the young-of-the-year (YOY) walleyes in Lake Mendota (Wisconsin, USA). The IBM was used to evaluate stocking strategies for walleye fingerlings. According to the rules of this simulation model, predation by a walleye would occur only if the walleye was sufficiently large relative to the prey individual. The length-frequency distribution of the YOY walleye population at the end of the 1989 growing season was predicted accurately by the model. During 1989, walleye fingerlings with a mean total length of 50 mm were stocked into Lake Mendota on 28 June. Simulations were performed to investigate the effects of the size of stocked fingerlings and the timing of stocking on subsequent YOY walleye growth. These stimulations revealed that if walleye fingerlings were stocked on 28 June, at an average total length of 60 mm rather than 50 mm, then the proportion of large (total length of XX 175 mm) fish in the YOY walleye population at the end of the growing season would have increased threefold over the observed proportion. Economic cost per large walleye was minimized when average total length at stocking was 62 mm. Stocking 50-mm walleye fingerlings on 14 June instead of 28 June resulted in a tripling of the percentage of large walleyes at the end of the growing season.

1.11 Simulation of perch (Perca fluviatilis L.) population dynamics in Lake Constance Buttiker and Staub 1992

The fishing of perch (Perca fluviatilis L.) is of major importance in the larger Swiss lakes. Perch populations in these lakes are characterized by high natural mortality and growth rates. To get a better understanding on how a perch population and its fishing yield are dependent upon management and fishing rules, a computer program for the simulation of perch populations was developed and tested. First results concerning the management of perch in Lake Constance (Switzerland) are presented. They show that the minimum legal mesh size of 32 mm for gillnets is a good policy and should not be increased.

1.12 Forecasting effects of harvest regulations and stocking of walleyes on prey fish communities in Lake Mendota, Wisconsin Johnson *et al.* 1992

Two commonly used simulation models were combined to assess the effects of stocking and harvest regulations on the consumption dynamics of walleyes Stizostedion vitreum in Lake Mendota, Wisconsin (USA). An age-structured population model was used to estimate the effects of five harvest scenarios on walleye population and fishery characteristics. Implementation of a 15-in minimum size limit resulted in increases in total yield (by weight), average weight of fish harvested, and walleye biomass remaining in the lake. Changes in walleye age structure resulting from various harvest scenarios were used as inputs to an energetics model to estimate how prey consumption by walleyes would vary under different harvest regulations. Simulations indicated that a stocking program that produced 8,000 yearling recruits annually would double walleye predation on planktivorous fish compared with predation estimated in 1987, when the study began. The modelling further indicated that a 15-in minimum size limit in conjunction with a stocking program would triple predation rates. This increase would be sufficient to reduce recruitment of yellow perch Perca flavescens in Lake Mendota in most years. Combining these two modelling techniques provides a framework for fishery managers to forecast how prey populations might respond to harvest regulations for gamefish.

2.1 Kootenay Lake Fertilization Response Model (KLFRM) Walters *et al.* 1991

The general aim of KLFRM is to simulate long term (30+ years) changes in plankton, kokanee, and Gerrard trout populations in response to changes in lake fertility associated with phosphorus loading. The model also tries to simulate effects of strategic fisheries management options such as varying exploitation rates and enhancement of spawning success through technologies such as spawning channels. The model is provided with historical inputs (nutrient loadings, water flow patterns, changes in kokanee spawning habitat) for the baseline simulation period 1960-89 so that its predictions can be compared with various data time series available for that period; it can extend the predictions out to the year 2010 using assumed input patterns provided by the model user.

The simulation program has five basic submodels for key components of the Kootenay Lake production system:

1) nutrient budgets (inflows, mixing processes, losses);
2) phytoplankton/grazing zooplankton production;
3) *Mysis* energetics and population dynamics;
4) kokanee energetics and population dynamics; and
5) Gerrard trout energetics and population dynamics.

The key use of the model has been to trace the impacts of experimental fertilization of Kootenay Lake through the pelagic food web. KLFRM had been used to identify gaps in our understanding of Kootenay Lake dynamics and to recommend research priorities. Model predictions suggest that fertilization will increase Mysis production and increase the competitive effects on kokanee production. However, given the considerable uncertainties identified by the authors around this prediction, an experimental fertilization program coupled with intensive monitoring was recommended, and is currently in its third year.

2.2 Carnation Creek Chum and Coho Salmon Models
Holtby and Scrivener 1989

The population dynamics of coho and chum salmon have been studied at Carnation Creek since 1970 as part of a multi-disciplinary study of the effects of logging on a small salmon stream in a coastal rainforest. The Carnation Creek salmon models predict the numbers of chum and coho salmon from correlative (empirical) relationships between survival and growth at various life stages and (1) climatic, hydrologic and physical variables, (2) indices of those features of stream habitat that were affected by logging, and (3) exploitation rates in the fishery. The models were used to partition the variability in adult returns between the effects of climatic variability in the stream and ocean, changes in stream conditions caused by logging and variations in fishing mortality.

2.3 Effects of sediment transport on survival of salmonid embryos in a natural stream: A simulation approach
Lisle and Lewis 1992

A model is presented that simulates the effects of streamflow and sediment transport on survival of salmonid embryos incubating in spawning gravels in a natural channel. Components of the model include a 6-yr streamflow record, an empirical bedload-transport function, a relation between transport and infiltration of sandy bedload into a gravel bed, effects of fine-sediment infiltration on gravel properties, and functions relating embryo survival to gravel properties. High-flow events drive temporal variations in survival; cross-channel variations in bedload transport cause spatial variations. Expected survival as a result, varies widely from year to year and between spawning runs in a single year. Alternative functions from previous research that relate survival to fine-sediment concentration in spawning gravel and to intergravel rates of flow yield categorically different results. The relative uncertainty of the components of this model indicates that the greatest research needs are to understand how sediment transport affects the intergravel environment and how these changes affect embryo development and survival.

2.4 COWFISH
Contor and Platts, 1991

COWFISH estimates monetary loss and fish loss due to livestock impacts based on seven habitat attributes by comparing estimates of optimum and existing numbers of catchable trout (> 6 inches) in a stream reach, regardless of species. Optimum number of catchable trout is defined as the number that would have been present if the stream had never been grazed by domestic livestock. Existing numbers of catchable trout are the numbers present under the current management situation. COWFISH then estimates the difference between the estimated optimum and estimated existing number of catchable trout. This difference is the estimated fish loss due to livestock use. To calculate total economic loss, COWFISH assigns $10.65 for each fish lost. Recreational loss is also estimated.

The seven input variables for the models empirical relationships are estimated ocularly. The inputs are:

1. % of undercut bank;
2. % of overhanging vegetation;
3. % of trampled vegetation or exposed soil on the bank;
4. % of riffle area covered by fine sediments;
5. stream width-depth ratio;
6. stream gradient; and
7. parent rock type

COWFISH is:

1. applicable throughout the Western U.S. (with adjustments);
2. useful any time there is not snow cover, but it is best if used immediately after the grazing season;
3. useful to evaluate large homogeneous stream sections (data collected from 5 different sites per stream mile are needed to provide a 10% sample of the study area);
4. suitable for streams less than 18 feet wide with low channel gradients, erodible banks, and grass-forb-sedge riparian areas; and
5. successfully used by untrained personnel using ocular estimations.

COWFISH limitations are that the:

1. accuracy of results diminishes when estimation of grazing impacts of fish production does not immediately follow the grazing season;
2. the model is less accurate for those streams with rocky streambanks, widths greater than 18 feet, channel gradients over 5%, and with forested riparian zones; and
3. the model outputs reflect population numbers for the immediate area sampled and not for the complete stream.

COWFISH appears to work satisfactorily in Montana for rainbow and cutthroat trout but performed poorly in validation systems within Idaho, Utah, and Nevada.

2.5 Atlantic Salmon Regional Acidification Model (ASRAM)
Korman *et al.* 1994

ASRAM is an age-structured salmonid life history model designed to examine alternative management (stocking, harvesting, liming) scenarios in Atlantic salmon rivers of the Maritimes. The model combines pH-dependent toxicity relationships, habitat-preference models, density-dependent processes, and natural and fishing mortality rates to predict annual smolt production and adult returns at site and river system levels. The spatial structure of the model is based on a database separating rivers into reaches where length, width and surface grade are approximated from orthophotographic maps and aerial photographs (available for 120 rivers in Nova Scotia). An empirical relationship between parr density and stream surface grade is used to distribute the initial number of fish for a simulation among reaches in the modelled river system. Annual pH time series collected from monitoring stations distributed throughout a river system are assigned to a set of spatially-linked reaches to predict tributary-specific pH-driven mortality rates. Relationships predicting parr size from density, and early marine survival from smolt size are the key density dependent functions which limit stream carrying capacity.

2.6 Relation between mortality of young walleye (Stizostedion vitreum) and recruitment with different forms of compensation.
Jensen 1992

The relation between mortality of young fish and recruitment is important for assessment of the environmental effects of facilities that kill large numbers of young fish, such as electric power stations and hydropower plants. A simulation model with a bioenergetic growth component was applied to examine the relation between mortality of young and recruitment of walleye (Stizostedion vitreum) with different forms of population regulation, including: food limited growth, food limited growth with size-dependent mortality, and food limited growth with age at maturity dependent on size. With food limited growth small increases in mortality of young reduced recruitment considerably, but the population slowly approached a new equilibrium. If mortality of young increased when growth was food limited, the population approached a new equilibrium of natality and mortality because with fewer individuals there was more food per individual, and individuals were larger in size and produced more eggs; this feedback adjusted natality to equal mortality. With either mortality or age at maturity dependent on size, large increases in mortality of young resulted in only small decreases in recruitment.

2.7 Potential impacts of global climate change on Pacific Northwest spring chinook salmon (Oncorhynchus tshawytscha): An exploratory case study
Chatters *et al.* 1991

Increases in atmospheric concentrations of greenhouse gases are predicted to raise global temperatures by up to 3 degree C over the next one-hundred years, which may have significant effects on natural resources. Even a smaller (2 degree C) temperature change may impact one prominent Pacific Northwest natural resource, the spring chinook salmon, Oncorhynchus tshawytscha. A computer model was developed by the Northwest Power Planning Council (NPPC) for use in developing the NPPC salmon enhancement plan for their Fish and Wildlife Program. Using this model, we investigate the impact of global warming on the production of spring chinook salmon in the Yakima subbasin of the Columbia River System. The model stimulates current prevailing environmental conditions and the implementation of improvements in salmon habitat planned by the NPPC. The data are then changed to reflect conditions that we infer to have existed between 6,000 and 8,000 years ago, when temperatures were approximately 2 degree C warmer than today. When the NPPC computer model is run under these altered conditions, it shows that projected climate change might reduce by half the Yakima River spring chinook salmon production predicted under both current and NPPC-improved conditions. These results strongly support the need for planned improvements in the fishery, since a 50% decline in existing fish populations could decrease spring chinook salmon abundance and possibly other salmonid populations beneath levels needed for the survival of the species. More broadly, the results suggest that if future global warming takes a form similar to that of 6,000 years ago, it could have major effects on the salmon population of the Pacific Northwest. Although some races of salmon might have their survival enhanced, others might be harmed. We recognize that all species and races would not be affected in the same way as Yakima River spring chinook, yet global warming is still a matter of concern because many of the Pacific Northwest salmon stocks are already under stress from other causes. A more comprehensive and thorough analysis is urgently needed.

2.8 A stochastic, compartmental model of the migration of juvenile anadromous salmonids in the Columbia River Basin
Lee 1991

A probabilistic model is developed which describes the juvenile migration of Pacific salmon and steelhead trout (Oncorhynchus spp.) in the Columbia River Basin in western North America. The downstream passage of smolts through a series of reservoirs or river reaches is modeled as an irreversible particle diffusion process through a series of compartments. The probability of live passage from one compartment to the next t time units after having entered the compartment can be viewed as the product of the probability of having survived to time t, the probability of transiting the compartment in t time units, and the probability of a successful transition between compartments at time t. From this basic premise, a general passage model is developed. Focusing on survival through Columbia and Snake River reservoirs (as opposed to dam-related mortality), a specific model which uses a Poisson death process and a gamma distribution to describe transmit times is presented. Analytical solutions to the single-compartment system are derived with extensions to the multi-compartment system. Empirical data from the Columbia River is used to demonstrate the utility of the model as well as to highlight problems that exist in fitting the model using existing data. Graphical techniques are presented for using the model to assess the effectiveness of management actions that are designed to improve the reservoir survival of smolts.

2.9 Integrating the Instream Flow Incremental Methodology with a population response model.
Cheslak and Jacobson 1990

The Instream Flow Incremental Methodology (IFIM) assumes that flow-dependent physical habitat and water temperature determine the carrying capacity of streams for fish. We have constructed a mechanistic simulation model that integrates the results of an IFIM Physical Habitat Simulation model (PHABSIM), a network hydrological model (NETWORK), an IFIM stream temperature model (SNTEMP), and field fishery studies to evaluate the potential effects of temporal changes in carrying capacities on a fishery resource. As an example of the methodology, the model was applied to an IFIM analysis of rainbow trout (Oncorhynchus mykiss) for a proposed hydroelectric project. Analysis of model performance indicates that it was well behaved, with reasonable, valid dynamics for each life stage and reach in the model structure. The model was used to compare population dynamics under natural and post-project conditions, and it indicated that a substantial enhancement in the fishery resource could be expected from instream flows proposed for the project. Inspection of reach-specific behavior was used to determine the basis of the predicted enhancement. These predictions are verifiable through a well-designed monitoring program, and a sensitivity analysis indicates that they are valid over a wide range of possible parameter values.

3.1 Functional response and capture timing in an individual-based model: Predation by northern squawfish (Ptychocheilus oregonensis) on juvenile salmonids in the Columbia River.

Petersen and Deangelis 1992

The behavior of individual northern squawfish (Ptychocheilus oregonensis) preying on juvenile salmonids was modeled to address questions about capture rate and the timing of prey captures (random versus contagious). Prey density, predator weight, prey weight, temperature, and diel feeding pattern were first incorporated into predation equations analogous to Holling Type 2 and Type 3 functional response models. Type 2 and Type 3 equations fit field data from the Columbia River (USA) equally well, and both models predicted predation rates on five of seven independent dates. Selecting a functional response type may be complicated by variable predation rates, analytical methods, and assumptions of the model equations. Using the Type 2 functional response, random versus contagious timing of prey capture was tested using two related models. In the simpler model, salmon captures were assumed to be controlled by a Poisson renewal process; in the second model, several salmon captures were assumed to occur during brief "feeding bouts", modeled with a compound Poisson process. Salmon captures by individual northern squawfish were clustered through time, rather than random, based on comparison of model simulations and field data. The contagious-feeding result suggests that salmonids may be encountered as patches or schools in the river.

3.2 Interactions between stochastic and deterministic processes in stream fish community assembly

Strange *et al.* 1993

Numerous studies have attempted to determine whether stream fish communities are structured primarily by deterministic or stochastic processes. Previous work has assumed that stream fish communities will show either persistence about an equilibrium because of strong density-dependent processes or random variation in structure as a result of environmental stochasticity. In a 10-year study of a California (USA) stream, fish community structure changed under the influence of storm-induced high discharge events that impacted recruitment. Species' relative abundances were altered as pre-recruitment stream discharges differentially influenced year-class strength among species with contrasting life histories. Simulation of stream fish community assembly under flow-driven recruitment variation indicates that community structure will vary depending on how particular high-flow events affect species' relative abundances and ongoing density-dependent processes, including competition and predation. Results suggest that stream fish communities are likely to show alternate states rather than a single persistent equilibrium. However, community assembly will not be random but will depend on situation-specific interactions between density-independent and density-dependent processes.

3.3 Predation and production by salmonine fishes in Lake Michigan, (USA) 1978-88 Stewart and Ibarra 1991

A marked decline of alewife in Lake Michigan during 1981-83 led to diet shifts by coho (Oncorhynchus kisutch) and chinook salmon (O. tshawytscha) from feeding primarily on large alewife to eating proportionately more immature alewives and other prey. Diets of lake trout (Salvelinus namaycush) did not change greatly during that period. Population biomass conversion efficiency averaged 24.5% for coho and 16.6% for lake trout. Chinook salmon suffered an apparent 20% decline in gross conversion efficiency of biomass (25.1 to 20.8%) and a 25% decline in average weight of sport-caught fish. We infer that chinook salmon growth was inhibited by insufficient forage available to them. A simulation of chinook salmon feeding on bloater (Coregonus hoyi) at 8 degree C suggested that such behavior could lead to further declines in growth rates. Extension of modelling results to include approximations for brown trout (Salmo trutta) and rainbow trout (O. mykiss) revealed peaks in total annual salmonine predation of 71,000 t in 1983 and 76,000 t in 1987. The alewife was 70% of all prey eaten by salmonines in 1987-88. Lakewide gross production by salmonines was 15,300 t (or 0.27 g/ m2) in 1987. Ratios of annual gross production to average monthly population biomass were 1.6 for chinook, 1.15 for coho, and only 0.6 for lake trout.

3.4 Individual-based model for growth of young-of-the-year walleye: A piece of the recruitment puzzle
Madenjian and Carpenter 1991

Young-of-the-year (YOY) walleye (Stizostedion vitreum vitreum) growth is a vital step in walleye recruitment. An individual-based model (IBM) was developed to describe the growth of YOY walleye in Oneida Lake (New York, USA) and Lake Mendota (Wisconsin, USA). In Oneida Lake the only prey species included in the model was yellow perch (Perca flavescens), whereas both yellow perch and bluegill (Lepomis macrochirus) were prey species in the model for Lake Mendota. IBM predictions for length frequencies of the YOY walleye population at the end of the growing season showed good agreement with observed length frequencies. A theoretical relationship was derived between the encounter rate, lambda, used in the IBM and the half-saturation constant, kappa, used in a type II functional response model. Estimates of kappa from the two models showed good agreement, thus corroborating the value of lambda chosen for the IBM application to Oneida Lake. The mean length of the YOY walleye cohort and the percentage of larger (175 mm in total length) walleyes in the cohort at the end of the growing season were most sensitive to gross growth efficiency, bioenergetics parameters for maximum daily consumption by walleyes, and the ratio of prey length to predator length at which the prey is susceptible to predation. In Lake Mendota the vulnerability of bluegills to predation by YOY walleyes was especially important in determining the growth of walleyes during their first growing season. The IBM approach was valuable for modelling those stages of life history in which characteristics of the individual were critical in determining recruitment.

3.5 Evaluation of alternative models of the coastal migration of adult Fraser River (British Columbia, Canada) sockeye salmon (Oncorhynchus nerka)
Pascual and Quinn 1991

A set of stochastic discrete step models of individual fish movement was developed to investigate the efficiency of compass orientation as a migratory mechanism in the coastal homeward migration of Fraser River sockeye salmon (Oncorhynchus nerka). Ultrasonic tracking data provided empirical values for the required parameters. Alternative movement models were validated by comparing the results of Monte Carlo simulations and known features of sockeye migration: general aspect of individual trajectories, timing, success in reaching the goal, and spatial occurrence. The effect of different headings and directional precisions on the probability of success in reaching the goal were considered. The more complex models captured the essence of observed movement patterns, but the number of fish getting to the goal was lower than expected. Many "modeled" fish were lost in the complex web of channels and inlets characteristic of this area. We conclude that the preference of a compass direction is not a sufficient mechanism to explain the observed migratory behavior of Fraser River sockeye salmon in coastal areas. Other mechanisms, such as negative kinetic responses to water from nonnatal rivers or short-term learning, may prevent the fish from being trapped in complex areas.

3.6 Biomass size spectrum of the Lake Michigan (USA) pelagic food web
Sprules *et al.* 1991

Biomass size spectra for the complete Lake Michigan pelagic food web from picoplankton to salmonids were constructed for nine sampling transects around the lake in May and in September 1987. Size spectra were typical for freshwater, having distinct peaks corresponding to major size groups. Biomass concentration of algae, zooplankton, and planktivores conformed to particle-size model predictions, but piscivore biomass was lower than predicted because these species are stocked. Mean annual total pelagic biomass was 72.3 g m-2 compared with a predicted range of 78.8-85.3 g m-2. Potential production of piscivores, Mysis, and Pontoporeia was in agreement with model predictions. No estimates of zooplankton or planktivore production were available, but we calculated that these could be 72.1-91.6 and 2.5-4.1 g m-2 yr-1, respectively. Our analyses suggest that piscivore production is constrained by food web structure. Bloater, which comprise 72% of planktivore biomass, make up less than 20% of salmon diets. We estimate that piscivore production could be double the current value of 0.27 g m-2 yr-1 if the forage fish community changed to include species more available to salmon.