Ecohydraulics: a new interdisciplinary frontier for CFD

By

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1. The river system in perspective

The *river continuum concept* states that the evolution of abiotic variables along a spatial gradient (upstream/downstream) is the main feature defining fish assemblages and distribution in streams (Vannote *et al.*, 1980). The notion of riverine habitat belongs to this conceptual framework. The biological functions of aquatic species and their communities are accomplished locally in a stream reach or river facies, but may vary in location in the hydrographic network depending on the spatial distribution of limiting abiotic factors and their relationship with hydrology and water use. The habitat requirements can include open sea or coastal zones for particular phases of the species life cycle. For most fish species, one can distinguish the following life phases and the corresponding habitats: *reproduction* and *egg incubation* in the spawning grounds, *fry emergence* and *feeding of the juveniles* (rearing habitats), *upstream and downstream migrations* often defined by the swimming capability of fish to pass through obstacles such as steep rapids, falls or civil works, *rest, predator avoidance* and *overwintering* which require some shelter habitat.

From a numerical hydraulics standpoint, the simulation of abiotic factors defining riverine habitats represents a fairly new challenge (Leclerc, 2002) because of the increased level of accuracy required to provide results at a local micro-habitat scale, corresponding to the instream flow needs of aquatic species or in studies of fish behaviour. The main variables traditionally considered to represent potential aquatic habitats are physical (velocity, stage, depth) and morphological (including riverbed composition). But distance to cover, hydraulic gradients, possibility of stranding, territoriality and competition are all factors that influence physical habitat quality and which can be investigated using CFD models Physico-chemical attributes of water can contribute to actualise or inhibit this potential. Again, study of such variables can benefit from CFD through transport-diffusion-kinetics models.

In this chapter, we will adopt a general perspective with regard to the ecological aspects of habitat modelling. We will try to demonstrate the conceptual framework and mathematical background defining the notion of habitat and focus on the new challenges offered to Computational Fluid Dynamics (CFD) practitioners, specifically those involved in environmental hydraulics. Even though the mathematical formulation of abiotic preferences of aquatic species and of habitat availability is fairly simple compared to the sophisticated algebraic systems defining the dynamics of the flow, several underlying hypotheses limit drastically the interpretation of results. The collaboration of knowledgeable ecologists with numerical modellers is absolutely essential to avoid mis-understanding of model outcomes.

Some of the proposed methods or tools used in microhabitat modelling methodology are fairly new, even in the academic context, and they still need some proofing in operational situations (e.g. development of habitat time series). For this reason, among others, this applied science has a very promising future for to the next generation of hydraulicians. Public policies will contribute significantly to the emergence and development of this methodology. For example, the European Water Framework Directive (WFD) requires catchment-wide habitat assessment. In the UK, habitat assessments are required for designing abstraction management plans. In the US, Environmental Impact assessment is the driver, especially in relation to the relicencing of hydroelectric power plants.

2. Conceptual modelling of habitats

Among existing types of models representing individuals, populations, communities, ecosystems and/or their habitats, one can identify: microhabitat modelling, population dynamics, bioenergetics, stock-recruitment, nutrient dynamics, individual-based ecological modelling, etc. The mathematical formulation of these models varies drastically from one option to another but some of them do not need a local representation of the flow domain (e.g. population dynamics) and consequently their mathematical framework does not rely on partial differential equation (PDE's) to take into account the distributed aspect of the flow field. However, some avenues relate to distributed variables, such as the Instream Flow Incremental methodology (IFIM) with its PHABSIM module (Physical Habitat Simulation) proposed by Bovee (1978) and Bovee and Milhous (1978). However, this approach is based on a simplistic definition of the flow based on traditional one-dimensional (1D) hydraulic tools originally developed for engineering purposes and mostly used to predict water levels and rough estimates of mean velocities. More recently, Leclerc *et al.* (1994; 1995; 1996) and several other contributors (see below) have started to make use of modern CFD and discretisation methods to provide more detailed flow fields to river ecologists. This improvement has gained in popularity in the last decade and it has become a strong component of the state-of-the-art in this domain (Leclerc *et al.,* 2003). Several chapters of this book present in detail all aspects related to CFD. In this chapter, we will focus on physical habitat modelling and aggregate formulations which suppose that fields of abiotic variables are available through field measurements and/or numerical simulation, and that these variables are discretized and can be numerically processed in terms of habitat values for each target species.

Habitat modelling attempts to predict the quality, useability or suitability of physical conditions but does not aim at predicting explicitly population dynamics or the presence/absence of species. Usually, habitat modelling does not take into account explicitly the ecological processes underlying the preference formulae defining the calculated values. Mostly based on the observation of presence or absence of the species in specific abiotic conditions, habitat models look more like “black boxes” than sophisticated conceptual tools (Leclerc, 2002). Thus, such models usually remain relatively simplistic and sometimes frustrating for river ecologists who often perceive them as highly reductionist engineering tools. In fact, several underlying hypothesis, unfortunately not always recognized and formulated, and rarely verified with proper validation protocols, can limit the predictive success of this methodology. Nevertheless, scientists who have used habitat models to improve their knowledge of rivers and related ecology would admit that such techniques also helped to introduce ecological considerations into the negotiation process of water resources allocation in a quantitative way.

2.1 The paradigm of numerical habitat modelling

The three main components of aquatic habitat models are (Figure 1):

1 one or more *living species* (fish, invertebrate, plant) with their specific abiotic preferences for each flow dependant phase of their life cycle,
2 a proper description of *environmental hydraulics* and of other abiotic factors contributing to the habitat occurrence,
3 a *drainage basin including water uses* and *allocation schemes* contributing not only a hydrological regime, natural or influenced, but also physico-chemical water attributes such as water quality, substrate grain size, temperature, nutrient loads and contaminants.

Distributed habitat models are the result of the post-treatment of the flow field (item 2) by making use of specific transfer functions defining the habitat *preferendum* (item 1). Taking into account the hydrological regime (item 3), whether influenced or natural, provides an essential relationship between habitat availability at the reach scale and catchment influences.

2.2 Modelling the aquatic species

2.2.1 Ecological role and selection of abiotic factors for habitat models

Abiotic factors can play several ecological roles during a life cycle and thus contribute to define the habitat value. For example, velocity exerts a very important influence at different phases of the life cycle of salmonids, and for several other fish species. For example, the summer feeding behaviour of juvenile salmon (0^+ to 2^+ years old) relies very much on the speed of the current carrying the food drift (invertebrates). The fish catches its prey by swimming against the current, often by climbing from its stand-still watching position into the recirculation zone located behind its “home rock”, through the turbulent boundary layer often up to the water surface. To achieve a net gain for its own growth, the total energy expenditure (including the basal metabolism) has to remain below the amount of energy contained in the prey.

Some debate can take place about which velocity to consider as a priority in habitat models: the *nose (or snout) velocity* or the *mean value over the water column*? This choice is highly relevant to the selection of an hydrodynamic modelling option, especially its discretization scheme (2D *versus* 3D). Some modellers argue that only 3D models can describe the vertical profile of the flow which would allow one to analyze accurately the fish behaviour, or specifically for understanding the fish bioenergetics (feeding and resting). Supporters of 2D models argue that the mean velocity is more integrated with respect to the feeding ecology of fish even though no direct or indirect description of the vertical velocity profile is provided by 2D tools. They also argue that even 3D models fail to take into account the random distribution of larger substrate elements and their transient effects (turbulence) on the flow patterns close to the bottom[1]. In fact, the nose velocity is a fish property rather than a flow characteristic. Moreover, this specific range of velocity occurs somewhere at the bottom of the turbulent boundary layer, and in practical decision making contexts, one does not really need to know the exact position where the fish can rest between two prey catches. However, if one seeks to develop detailed bioenergetic versions of habitat model (e.g. individual based ecological modelling) such a detailed 3D representation might become necessary.

[1] A section of this chapter (2.3.5) proposes an interpretation of the inherent flow complexities that can contribute to habitat occurrence in rivers and the abilities of hydrodynamic models to represent them.

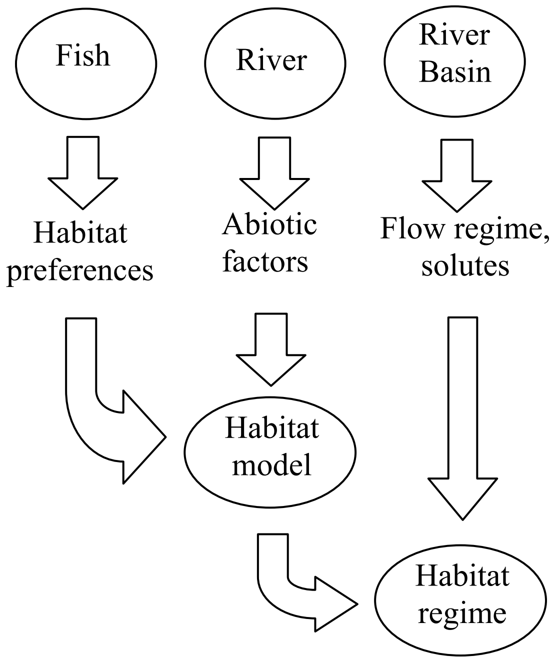

Figure 1 : The paradigm of habitat modelling (After Leclerc, 2002)

Similar debates could also be mentioned for substrate grain size and composition which are also very important for feeding because mid-size rocks and stones provide very local flow features (microhabitats) which allow the fish to rest between its attacks on prey. Moreover, spawning activity, egg incubation, predator avoidance and fish over-wintering (notably, in cold climates) are partly governed by the grain size and composition of substrate elements which provide shelter between interstices and thus, oxygenation of interstitial waters. Granulometric distribution of substrates is also important in order to avoid the presence of fine materials which result in the filling up of interstices, and occasionally, the deleterious paving of the riverbed. This is not only harmful for reproduction activity, but also for feeding of fish because the clogging of substrates restricts the production of benthic invertebrates that represent the main food source for many fish species (Milhous, 1996).

Traditionally, microhabitats of fish or other species living in river are mostly modelled by three key abiotic variables: mean velocity of the water column, depth, and riverbed substrate (mean diameter) (Bovee, 1978). These generic variables can be considered as the basic template of aquatic habitats: velocity mainly determines the bioenergetics of individuals through feeding and swimming; substrate influences the shelter opportunities, the production of invertebrates and the quality of spawning grounds; depth relates to light penetration and may influence the predation process.

Other important physical and/or physico-chemical attributes, such as water temperature, availability of light in the water column or close to the riverbed, or turbidity can also play an important role and must often be considered (Bechara *et al.*, 2003). The slope of the riverbed and the presence of tree cover are other examples of variables that may be measured or modelled for a more thorough assessment (Morin *et al.*, 2003; Hardy & Addley, 2003). In the case of larger fluvial systems with long wind fetches, wave climate can also become an important habitat variable, as can the organic content of sediments (Morin *et al*, 2003).

Lamouroux and Souchon (2002) have proposed aggregated abiotic variables as the main explanatory factors of habitat distribution over the river network (mesohabitat). For a given segment, the governing processes belong to the flow regime defined by the mean Froude number of the reach, the average rate of flow per unit of width (specific discharge) and the median grain size. The Froude number reflects the lentic (slow currents)/lotic (fast currents) character of the flow regime which obviously connects strongly with distinct ecological strategies of fish species. As to the specific discharge, this variable aggregates in one single term the combined effects of velocity and depth, both dominant factors of the standard micro-habitat. Mean substrate size obviously reflects the role of this variable at the local microhabitat scale (shelter, home rock, feeding position).

2.2.2 Defining the preference range for abiotic factors among species

The classic approach (Figure 2) to quantifying habitat value consists of estimating local habitat indices (so-called “preference curves”) based on a knowledge of the optimum ranges of abiotic conditions (the so-called *preferendum*) for the targeted species during a specific life stage (Bovee, 1982). In a river, the life stages considered often relate to those potentially influenced by variations of the flow regime (Bovee, 1978). A Habitat Suitability Index (*HSI* or Global index) is thus calculated by assigning a weight (in the form of a power factor), between 0 and 1, to each abiotic variable defining habitat, and combining these values in a global index, usually by way of a simple geometric or weighted geometric mean (Equation 1). The geometric mean is often used because it implies that each independent value can play the role of a limiting factor (i.e. if the value of one independent variable is 0, *HSI* = 0). If the local index for a given variable is one, than the requirement is completely met for the target species. Due to its great simplicity, such an algebraic formulation must be interpreted with caution. Further explanations with regard to this will be provided later (section 2.2.3).

$$HSI(x,y)=I_V^a.I_H^b.I_S^c \quad \text{with} \quad a+b+c=1.0 \tag{1}$$

Where: I_V, I_H and I_S represent basic indices for velocity (V), depth (H) and substrate (S) respectively and a, b, c are weights powering the basic indices and influencing the relative ecological importance given to each basic index in the model.

Most often, the weights are assumed to have an equal influence upon each of the factors considered (a=b=c=1/3). In some cases, statistical techniques such as principal components methods or regression can be employed to discriminate the relative explanatory importance of each abiotic factor upon the fish distribution. The percent of residual explanation of each variable can then be used as the weighting factor (Leclerc *et al.* 1995). However, as the mentioned variables are obviously auto-correlated through the logarithmic representation of the velocity profile forming the turbulent boundary layer, one can not be sure of the adequacy of this approach and this criticism has often been mentioned as a drawback of this modelling approach (Shirvell, 1986).

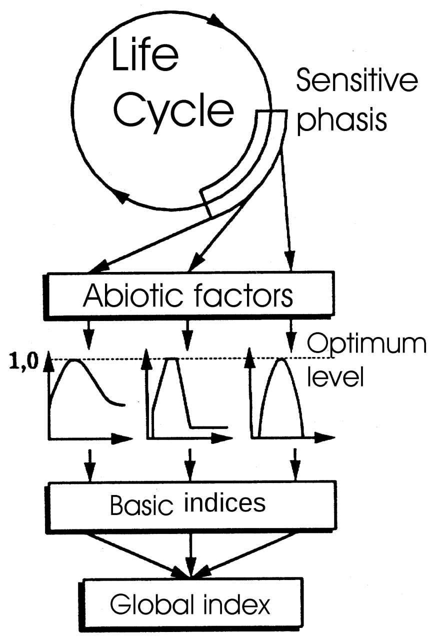

Figure 2 : Building a classical fish *preferendum* model

Each basic index forming the *HSI* formulation is represented by preference curves. They can be built for the range of abiotic conditions encountered, based on field observation data of the local fish presence or absence in the stream, and on the value of controlling abiotic factors measured at the corresponding location. Fish observations are realised by snorkelling or electro-fishing means (Schneider and Jorde, 2003). In some situations where observation is restricted by turbidity, sediment load or water colour, expert advice can replace measurements. A mixed measurement-expert approach can also be mobilized for adjusting empirical curves, especially in the extreme ranges of physical variables, and used to obtain more meaningful results with respect to effective habitat value.

When building the preference curves (or basic indices), one must take into account the availability of the entire range of flow conditions within the flow domain. More frequent conditions might artificially appear preferable if very abundant. If the statistical method used to determine the *preferendum* takes into account only the conditions where the presence of fish was observed, one obtains a "utilization curve" which is still a useful piece of ecological information but does not strictly represent the preferendum.

With field data available, the most common methodology (Figure 3) to establish the *preferendum* consists firstly of building histograms of the presence of fish as a function of the abiotic variables observed at the fish location. A second histogram representing the total number of presences and absences of fish with respect to local abiotic conditions is also prepared. The later histogram is considered to represent the availability of conditions. The *preferendum* is obtained by dividing the first histogram by the second (classwise) and normalizing the result by setting the maximum value to 1.0.

In the hypothetical situation proposed in Figure 3, one can observe that the maximum number of fish presences occurs in class 3 of the independent variable while the maximum availability of abiotic conditions occurs in class 4. It is also interesting to observe that class 2 offers half of the amount of river space available in class 4 but the number of fish presences in class 2 is superior to class 4 which means a greater fish density, and consequently more favourable habitats. As already stated, in Figure 3, a normalized graph of the fish presence histogram would be interpreted as a "utilisation curve", which, when employed instead of the preference curve, could lead to inaccurate results.

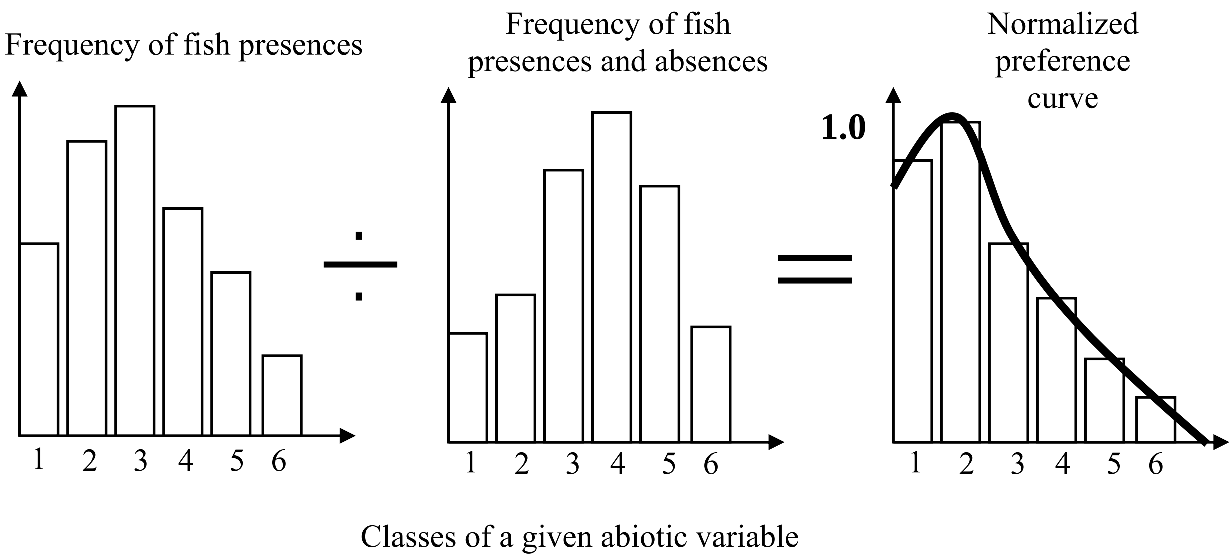

Figure 3 : The general methodology for establishing fish preference curves

One of the main criticisms of this classic *HSI* approach is that it does not take into account the fact that the so-called independent variables are not truly independent (e.g. flow and depth). The information content provided by some of the abiotic variables may be redundant.

In spite of its drawbacks, the *HSI* approach is still commonly used and can often be validated by the measurement of the density distribution of the target species in the field (e.g. Guay *et al*., 2000). However, it remains difficult to transfer this local modelling parameterization to other rivers.

2.2.3 The interpretation of preference curves

Let's look more carefully at the interpretation of the algebraic formulation of *HSI*, specifically when considering the combination of a geometric mean and basic indices defined in the range of 0 to 1. As already mentioned, a zero value for a given basic index produces a zero *HSI* value as a result, which means that this index plays the ecological role of a limiting factor. A value of 1.0 for a basic index does not influence the *HSI* value, which means that local conditions comply with the fish abiotic preferences. These statements can play an important role when building preference curves. If no or a very small number of fish are observed in a given class of abiotic variable, does this mean that this range is restrictive for the presence of fish? Possibly not. For example, the river reach might not be occupied optimally by the fish population considered. In turn, this could be related to the presence of predators, a low population with respect to the current habitat availability during the field characterization, or to adverse factors such as the presence of contaminants inhibiting biological functions.

Moreover, the habitat availability for this particular phase of the fish life cycle may not be the limiting factor for the population size.

Consequently, it is recommended that one adapts the preference curves obtained from the standard method in the extreme ranges of the variables by taking into account the existing ecological knowledge of the species under study. Expert advice is recommended in this case. Ending the curve at an intermediate value (e.g. 0.3-0.5) could be acceptable in order to keep these areas under consideration, especially if all other basic indices behave satisfactorily.

2.2.4 Defining habitat value with a probabilistic approach

Probabilistic multivariate approaches such as logistic regression have recently been described in the literature and offer better possibilities for model validation (see section 3) and could even allow for better model transferability from one river to another (Guay *et al.*, 2000, 2002; Boisclair, 2003; Parasiewicz, 2003). Logistic regression can be used to model dichotomous variables (presence-absence) as a function of independent variables. If independent variables are standardized, logistic regression coefficients can be compared to identify which variables causes the greatest increase or decrease in the odds ratio. The protocol for physical habitat characterization is essentially the same in this case as for *HSI* and preference curves. A Probabilistic Habitat Index (*PHI*), which refers to the probability of presence of individuals of the target species can thus be calculated. These results can then be used to assess micro-habitat or meso-habitat (reach value, see Parasiewicz, 2003).

$$PHI(x,y,Q)=\frac{1}{1+e^{-P(V,H,S)}} \tag{2}$$

Where: P is a polynomial composed of linear and/or quadratic terms that are functions of controlling abiotic variables V, H, S, etc. where V is the velocity, H the depth and S the substrate grain size.

If the application of logistic regression appears to give promising results (Guay *et al.,* 2002), it must however be kept in mind that the interpretation of the regression coefficients is different than that of linear Ordinary Least Squares (OLS) models. Moreover, it must be noted that the coefficients of determination used to assess the goodness of fit of logistic regression models are also different from those of OLS models. The so-called pseudo-R^2 metrics are based on log-likelihood ratios (e.g. McFadden R^2) and are not the equivalent of the OLS R^2 used in linear regression models (Scheiner and Gurevitch, 2001).

The order (linear and/or quadratic) of the algebraic terms forming the polynomial P can have some importance for the predictive power of the model. The linear term often establishes a certain proportionality between the abiotic variable considered (independent variable) and the probability of species presence, especially in the lower range of the variable. The addition of quadratic terms as a function of the same variables transforms the polynomial P into a parabola which will damp the ever increasing or decreasing effects of the linear terms. This statement has a corollary regarding the range of value of explanatory variables considered by the measurement protocol. If this range is limited to a narrow domain of representativeness, the linear component might be sufficient to reproduce accurately the fish distribution in this limited region but with low predictability outside this specific range.

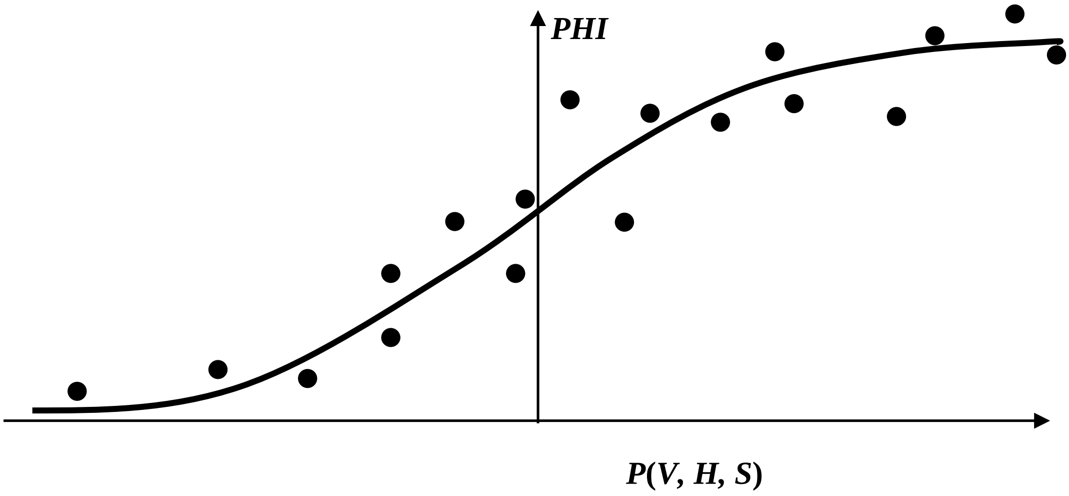

Figure 4: The logistic regression approach to the aquatic species *preferendum*

2.2.5 Fuzzy logic for modelling habitat *preferendum*

Alternate approaches have also been proposed to model fish preference, based on fuzzy or imprecise information available on the selective behaviour of target species (Schneider and Jorde 2003 ; Jorde *et al.,* 2001). Unlike previous methods that do not really allow one to consider the existing knowledge being offered by fisheries biologists, the fuzzy-rule based method operates with combinations of qualitative and semi-quantitative criteria to which a suitability level is attributed according to specialists who try to achieve a consensual interpretation. The procedure starts by setting up check-lists with possible combinations of relevant physical criteria and lets the specialists define in natural language whether habitat quality is low, medium or good for each of the proposed combinations. The basic information used is often classified by using common language (low, medium, fast velocity). The number of combination depends upon the number of variables considered, *J,* and the number of linguistic classes, k_j , defining each of them.

$$N=\prod_{j=1}^{J} k_j \tag{3}$$

In order to take into account the imprecise definition of the intervals of classification, fuzzy logic uses overlapping “membership functions” allowing one to classify a given variable value in a combination of two or more intervals of definition. The numerical habitat value being associated with a combination of basic explanatory variables takes into account the value of the membership functions which serve further as weighting factors. Thus, in practice, a given value of velocity can be partly low (e.g. 0.25) but mostly medium (0.75).

For further explanations about the numerical implementation of this method, the reader is advised to consult Schneider and Jorde (2003). This method has proved to perform better at the validation step than the standard *preferendum* method. It is fully implemented in a software tool called CASIMIR (Computer Assisted Simulation Model for Instream Flow Requirements) which forms an integrated toolbox for habitat simulation in rivers.

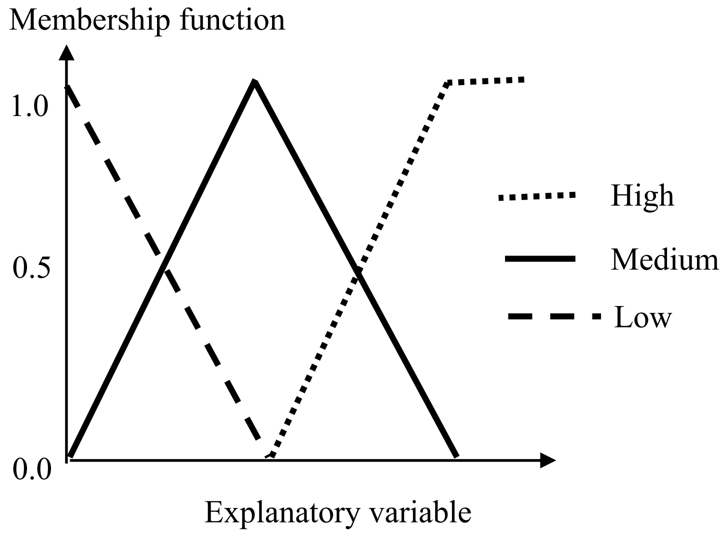

Figure 5 : Illustration of fuzzy delineation of habitat variable classes
(Adapted from Schneider and Jorde, 2003)

2.2.6 Guilds: aggregating fish *preferendum* models for similar species

Most habitat preference models have been developed for single species and these models are considered specific to a particular river with low transferability to other, even similar rivers. By combining data sets collected on different, but similar, river systems, it is possible to build more generic models offering a better representation of the entire range of preference of the target species. It is even possible to combine data sets formed by different species sharing similar ecological behaviour and niches. For example, Lamouroux and Souchon (2002) have suggested the use of *guilds*, a well-known ecological concept that allows for the combination of a number of species and even of various life stages that have similar habitat requirements. A multi-specific habitat model can thus be built. Grouping criteria are generally based on the assumption that those species share common feeding or reproduction strategies. Guild *preferenda* can be associated with more general reach characteristics such as riffle, runs, flats and pools (Lamouroux and Souchon, 2002). This approach may also allow for inter-basin transfer of *preferendum* parameters. This gain is counter-balanced by a loss of precision for a given species within a guild. The main advantage of this transferability is that general guild preference curves can be used in sites with scarce local data.

2.3 Characterizing and/or simulating abiotic factors controlling habitats

Data acquisition for some of the abiotic factors describing habitats is possible *via* field surveys, but in many cases, it is not feasible to do so and simulation model outputs are used instead (e.g. detailed spatial velocity fields, temperature distribution, wave climate in larger systems). In fact, a combination of field data and simulation results is mostly employed. For example, a hydrodynamic simulation requires field measurements such as the substrate composition (roughness), discharge values, stage-discharge relationships as boundary conditions, and topography to feed into the model.

2.3.1 The classical One-Dimensional (1D) approaches

One-Dimensional (1D) or semi Two-Dimensional (2D) models of the PHABSIM type (Bovee, 1978, 1982; Bovee and Milhous 1978; Bovee *et al.*, 1998; EVHA simulator by Ginot *et al.*, 1998) have been and are still used by specialists. This type of model, which constitutes the most popular module of the IFIM (Instream Flow Incremental Methodology proposed by the US Geological Survey) modelling system, is mainly characterized by field measurements at a limited number of sampling points spatially organized on transects (Figure 6). Each sampling point is centred on a rectangular cell and this discretisation scheme forms a structured grid covering the entire flow domain. This approach aims to represent the global variability of the morphology and the flow characteristics (depth, velocity) over a flow domain. Typically, a minimum number of transects (4-6) is necessary to capture a minimum number of different river *facies* within a given reach (succession of pools and riffles, meanders). The position of sampling points requires exact location in order to ensure some positional consistency between flow events. Whilst this logistic requirement does not represent a severe constraint for small shallow streams, it becomes seriously limiting for large river systems. Moreover, the riverbed morphology is only described on the transects so the approach tends to over-represent the lateral variability of the riverbed and under-represent it longitudinally (Ghanem *et al*, 1994; Secretan *et al,* 2001).

The flow field itself is empirically characterized at 5-6 different discharge values representing as much as possible the hydrological range relevant for the study. The PHABSIM approach, jointly with the measurements, allows one to employ simple semi-empirical flow models using 1D solution schemes ((e.g. Chezy-Manning-Strickler formula, HEC-2, HEC-RAS) to represent the flow field in a more predictive manner. However, the limited number of physical processes included in such models (mainly the gravity/slope and bed resistance to flow) do not allow one to extrapolate accurately outside the flow range covered by the characterization protocol. Calibrating such models can be troublesome if one considers that the cross-section based friction factor can evolve very significantly with the lateral extent of the flow domain (wetted perimeter) which depends on the discharge (or tide). Consequently, the calibration of such models needs to be revisited for different flow discharges or water levels. Another problem is raised by the requirement to gather homogeneous flow data sets over the entire flow domain with respect to discharge. This constraint is inherent to the model calibration process. For small catchments with highly variable hydrology, the discharge can change very significantly over a short period of time, and consequently jeopardize the measurement effort for the target discharge event.

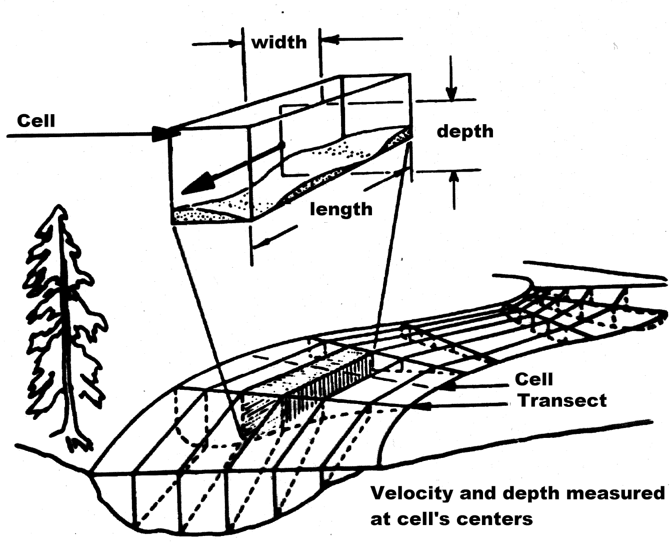

Figure 6: Curvilinear 1D PHABSIM discretisation scheme for characterizing abiotic factors

Moreover, the low frequency of extreme events often makes it difficult to build flow models covering a wide flow range, especially for droughts. For floods, the problem is rather related to logistic and safety constraints (e.g. access to site, high velocities). These considerations together with the limits of validity of discharge dependent friction factors seriously limit the range of extrapolation of such models. For very low droughts, this limitation can be troublesome when one aims to establish minimum conservation flow recommendations. Hopefully, habitat quality is often poor at extreme high or low flows, as defined by 'bell' shaped *HSI*s. Therefore habitat predictions at these discharges are less dependent on predictions of depth and velocity.

Additional criticisms relate to complex flow structures such as return flows behind obstacles or transient behaviour associated with tides or flash floods that can not be represented with such models. Moreover, potential modifications or alterations of the river morphology formed by enhancement works or natural river evolution cannot be modelled either. Considering that simple 1D approaches are highly dependent upon direct measurements of the current topography and flow conditions and provide low predictability, their utilization is restricted to flow domains that will not experience such morphological alterations.

One of the main reasons explaining the sustained utilization of this approach for more than 25 years, notably in the USA, is related to the legal role played by this model scheme and its necessary standardization. Most of the time, the results are used in the resolution of water allocation conflicts, in particular between ecological instream flow needs and other water resource utilisations, such as flow diversion for irrigation, drinking water supply or hydroelectricity purposes (Leclerc *et al.* 1995). Unfortunately, the high standardization level of the method severely restricted its adaptation and retarded the utilization of more sophisticated and accurate hydrodynamic tools currently available (at least 2D). Another reason could be provided by disciplinary barriers: even though the developers of IFIM (Bovee and Milhous, 1978) intended to elaborate an ambitious and comprehensive modelling system including water quality, this pioneering effort in habitat modelling was mainly led by non-hydraulic researchers who focussed as a priority on the development of the environmental and ecological aspects of the methodology with less consideration for the riverbed morphology and hydraulic representation. Finally, one must admit that, in the late seventies, very few 2D tools for modelling flow fields (considering drying-wetting processes) and visualizing results were then available and their use was seriously limited by the existent computer capabilities.

2.3.2 Two-dimensional modelling of abiotic factors

With the fast development of personal computing and Computational Fluids Dynamics (CFD), full discretised 2D hydrodynamic simulation tools have become more popular over the last decade because of their enhanced description of complex hydraulic features, especially for high velocity areas, rapidly varied flow regimes and back eddies (Leclerc *et al.*, 1995; Ghanem *et al.*, 1996; Hardy, 1998; Katopodis, 2003; Hardy and Addley, 2003; Secretan *et al.*, 2001; Heniche *et al.*, 1999; Waddle *et al.*, 1997). Finite elements, differences or volumes form distinct but similar discretisation schemes allowing one to represent numerically both the field data (Digital Terrain Model) and the mathematical framework describing the flow equilibrium (conservation of mass and momentum). However, the non-structured aspect of the finite element or volume options is reputed to offer a better adaptability to local flow and river bed features characterized by higher variable gradients, and consequently more accurate results.

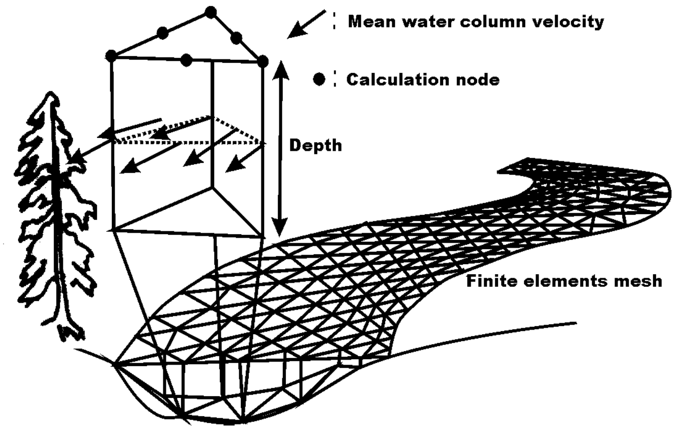

Figure 7: Finite element discretisation scheme in 2D habitat studies (After Leclerc, 2002)

As a 2D methodology necessitates more detailed field data, this requirement is sometimes perceived as a limitation for practical use because of the additional costs related to extensive field characterization strategies, in particular for topography and riverbed substrate. Nevertheless, the need for direct flow measurements (velocity, stage) is considerably reduced -compared to 1D approaches- considering that this data forms a limited sample of the flow characteristics, only necessary to calibrate and validate model predictions, and that it does not aim to represent the entire flow field as for 1D models. For the same reasons, the field campaign related to instream flow variables is reduced to typically two relatively distinct reference discharge events. This specification allows one to calibrate and validate the model on independent data sets and preserve to a certain extent the predictive power of the model outside the range of its establishment. In very particular contexts where additional forcing factors (e.g. seasonal variation of friction factors such as aquatic plants or ice cover) make the parameterization of the model more complex, more than two flow events representing a wider range of conditions can be necessary (Morin and Leclerc, 1998).

2D hydrodynamic models also allow one to simulate, and hence, to analyse the physical impact of important modifications in river morphology or riverbed substrate which are among the main basic factors limiting the local habitat value. Transient behaviour of flow is naturally taken into account in the Saint-Venant (shallow water equations) scheme. Such models also open the door to the application of sophisticated transport-diffusion simulation models which allow one to take into account a broader range of abiotic variables such as sediment transport, propagation and the fate of contaminants, thermal regime and so on. 2D simulation results also offer a much greater potential for computer graphics and meaningful interpretations, especially when coupled with Geographical Information Systems (GIS).

The implementation of these more sophisticated hydrodynamic models together with proper GIS tools for spatial analysis can be associated with more elaborate biological models that may, for instance, include key behavioural information, such as inter- or intra-specific competition or predator-prey avoidance relationships within the ecosystem (see Hardy and Addley, 2003). Distance to cover, exclusion zones dictated by the presence of a predator species and hierarchical data can be incorporated in a study scheme when multi-dimensional hydraulic models are used. Even more sophisticated biological models may include the period of re-colonisation by invertebrates of an area previously dried because of lower flows associated with dam operations. Other models may include bioenergetic components associated with feeding patterns or community characterization. This level of ecological sophistication is only possible if the habitat models are based on multi-dimensional (i.e. ≥2-D) hydraulics. Implementing concepts of discontinuous spatial distribution of habitats such as their "patchiness" (Bovee, 1996) necessitates accurate distributed results allowing the calculation of gradients or an account of discontinuities.

In the last decade, ever increasing desktop computing capabilities, new sophisticated data acquisition systems coupled with precise positioning such as the Global Positioning System (GPS), micro-photogrammetry with the assistance of low flight aerial photos or laser scanning, remote sensing using satellite or airborne imagery, and better current measurements using Acoustic Doppler Current Profiler (ADCP) technology have also contributed to use of new generations of hydraulic models to their full potential (Hardy & Addley, 2003).

2.3.3 Time for three-dimensional modelling?

In the future, three-dimensional tools are likely to play a growing role in representing even more complex flow situations in the context of habitat studies (Leclerc, 2002). For example, helicoidal (secondary) flows in meanders, vertically stratified flows associated with thermal regime or salinity, wind-driven flows in coastal areas can be represented accurately only by such models. However, transient behaviour associated with short period and local turbulence or complex flow patterns related to large substrate elements still remains beyond the current capabilities of most existing 3D models. Thus, it is very important to verify the underlying hypotheses posed to develop a particular model aimed at representing flows in a 3D fashion. These hypotheses can seriously limit their applicability, especially for habitat modelling purposes. For example, the incompressibility hypothesis limits the application to homogeneous water masses with respect to density. As with 2D models, hydrostatic pressure also restricts, to a certain extent, their application to rapidly varying flows. Finally, the extensive application of 3D codes to habitat modelling still needs corresponding empirical models (transfer functions) to describe species preferences in a 3D framework.

2.3.4 The need for drying-wetting capabilities

Wetting and drying is an important factor for assessing habitat (or ecological modelling) for many species, particularly plants. In this respect, there is great potential for CFD models to provide information on the timing and extent of inundation. Different factors can contribute to define the spatial extension of the wetted part of a given flow domain: tides, hydrological regime, and to a lesser extent, the magnitude and direction of winds. Thus, modelling flow variables with the shallow water equations, whatever the discretisation scheme retained, requires non-linear *drying-wetting* capabilities to deal with transiently moving boundaries. This problem is addressed in Bates and Horritt (2004; this volume). Heniche *et al.* (2002; 1999) and Leclerc *et al.* (1990a,b) also presented reviews of different approaches to solve this problem. Among several others, Hervouet *et al.* (1992) and BOSS Int. (2003) propose drying-wetting simulators that can be readily used for habitat studies. The accuracy of the drying-wetting solution scheme may become an important issue as the method needs to solve the flow equations in partly wet cells (or elements) of the solution mesh (near shores). In specific situations where biological functions take place in such areas, accurate values of flow variables are necessary to represent such habitats properly. As the solution scheme may be weaker in these zones, due to the partial information taken into account by the model (dry nodes do not really contribute to the solution scheme), one should remain careful when interpreting such results.

2.3.5 Limited extent of modelling capabilities for secondary and turbulent flows

Whilst modern hydrodynamic simulators offer ever enhanced capabilities to represent complex flows, research/development is still needed to represent accurately the whole range of flow features present in natural rivers, especially with regards to turbulence and secondary currents related to spatial heterogeneity of river morphology and alluvial composition. If modelling of hydraulic characteristics around large substrate elements is theoretically possible for single grain elements (e.g. boulders), it remains obviously impracticable for entire flow reaches due to limited computing power and to the unavailability of appropriate boundary conditions. Upscaling is also an important aspect that limits this type of modelling.

One can represent the entire range of variability of river flows with the following equation where the different terms proposed are defined in Table 1. The development of mass and momentum equations with different space and time integration schemes leads to more or less reduced representations of the flow. However, most of these local flow patterns occur at scales of ecological importance and they play a critical role in the occurrence of microhabitats. Unfortunately, most hydrodynamic modelling schemes still fail to simulate such complexities. Does this mean that modelling microhabitats, even with the most sophisticated 3D tools, cannot offer any useful information for decision making or for developing knowledge? Certainly not. But the need for more and more detailed results in the context of habitat studies will become an additional motivation for the future development and implementation of more elaborate simulation tools.

$$V(x,y,z,t)=\overline{V}(x,y,z,t)+V'(t)+V''(z)+V'''(x,y,z)+V''''(x,y) \qquad (4)$$

2.4 Digital terrain modelling (DTM)

2.4.1 Is measuring already a modelling process?

As stated before, input data for habitat models is provided either by predictive hydraulic models, or by field measurements as represented by a DTM, but mostly by a mixed combination of both. The purpose of digital terrain modelling is similar and complementary to hydraulic modelling. Its main goal is to homogenise and represent the field data sets necessary to simulate hydraulic variables, and sometimes, other flow attributes such as water quality, temperature or sediment transport. The DTM can also provide input data directly to habitat models (e.g. riverbed substrate grain size and composition). Strictly speaking, terrain models are limited to measured field data. Important variables that relate to the DTM include topography, substrate attributes, and stage/discharge relationships at open boundaries. In addition to these variables, one can also take into account any given independent factor, natural or artificial (man made), that influences the flow pattern such as the seasonal distribution of aquatic plants, debris accumulation, river engineering works and ice cover thickness and roughness.

Table 1: Spatio-temporal variability of flows of ecological relevance and hydrodynamic modelling schemes

Processes	**Variable notation**	**Description**	**Remarks**
Mean velocity	$\overline{V}(x,y,z,t)$	Mean term time dependant	Obtained by time integration of turbulent components of flow
Turbulence	$V'(t)$	Short term time dependant	Accounted for in momentum equations by turbulent shear stresses (turbulence closures) but rarely represented
Vertical profile	$V''(z)$	Turbulent boundary layer; theoretically logarithmic	Accounted for by bottom resistance (1D-2D) or explicitly represented by 3D models
Secondary flows	$V'''(x,y,z)$	Helicoidal meandering flow	Only explicitly represented in 3D (but allowed for in some 2D models)
Tertiary flows	$V''''(x,y,S)$	Local scale dominant; substrate grain size dependant	Hardly accounted for by transient turbulence closures (horizontal shear stresses), rarely simulated, even in 3D

The choice of a proper field characterization strategy is very important in order to allow accurate application of predictive models. In fact, terrain modelling starts immediately with the field work and our belief is that "*Measuring is already modelling*" (Leclerc, 2002). For example, characterising the riverbed topography with lateral transects as is usual for 1D hydraulic models is a strategy that provides good lateral accuracy, but a very poor longitudinal description of the shapes and flow features in between.

A 2D random and/or adaptive distribution of measurement points based on local interpretation of the variability during the field campaign leads to a far better representation of river morphology, and consequently, more accurate results with respect to habitat variables. Nowadays, modern instrumentation for measuring topography and other physical variables coupled with accurate *x,y,z* positioning allows one to build a very accurate DTM for a fraction of the cost of older methods. This type of relatively coarse representation of the actual topography can be seen as a model.

The substrate grain size distribution plays a very important role either for parameterising flow resistance from the river bed, or computing local habitat preference values. For flow resistance, local *x,y* properties of substrate roughness, and macro-roughness in situations where sediment is mobilised and transported by the flow are necessary information to account for in a DTM underlying 2D or 3D hydraulic models. Several empirical formulae of the Manning-Chézy-Strickler type have been proposed in the literature to parameterise substrate characteristic size into roughness coefficients.

On the other hand, substrate information for habitat modelling may need to be represented in a particular format more adapted to model the instream flow needs of the fish or plant species under study. For example, some habitat models may require in addition to the grain size, the organic matter content of the sediment (e.g. aquatic plants). Some fish species avoid fine particles to spawn and seek clean and coarse substrates allowing oxygenation of the eggs during their incubation. Most salmonids require fairly large substrates (pebbles, cobbles, small boulders) as “home rocks” during their feeding period, etc.. Thus, a characterization protocol with respect to substrate should simultaneously take into account the needs for hydraulic modelling and those of the habitat models to be developed.

2.4.2 The problem of heterogeneity of data sets

Most physical factors can be measured using several instruments with variable precision and spatial distribution schemes. For topography, characterization with total stations, echosounders coupled with Differential GPS (DGPS) positioning systems, or airborne remote sensing using photogrammetry or laser scanners can all provide good representations of limited parts of the riverbed shape (Goodwin and Hardy, 1999). However, most of the time, their intrinsic limitations (e.g., echosounding is only possible where a minimum depth exists) impose a need to mobilize more than one tool to achieve a complete characterization, and consequently, this generates more than one type of data. Moreover, these are sometimes heterogeneous with respect to their accuracy, their spatial distribution scheme, or worst, their georeference.

Moreover, the density and spatial distribution of data points depends on the targeted variable with, as a consequence, heterogeneous data layers that can not automatically be modelled in combination without some transfer procedures on a common compatible digital support, usually a mesh or a grid (Secretan & Leclerc, 1998; Secretan *et al.,* 2001), whatever the discretisation scheme retained (finite difference, element or volume). This so-called “receiving” or “client” mesh is usually the one utilised for hydrodynamic simulations. Thus, geomatic tools adapted to deal with heterogeneity usually allow one to edit the data points and interpolate between them in order to homogenise data sets by a proper transfer onto the compatible mesh. Usually, a continuous representation of most field data sets is initially obtained by using the Delauney algorithm (Baker, 1987) which allows one to connect the data points together in a single unique way called a TIN (Triangular Irregular Network), which is in fact a linear finite element grid. These meshes are called “server meshes”.

2.4.3 Validating the DTM?

Simulating flow fields with incorrect topographic data illustrates very well the expression "*garbage in, garbage out*". Error can arise either from inaccurate characterization strategies, or when modelling the field data (DTM). For this reason, it is preferable, if not essential, that the original data be preserved intact in the database and kept available for the subsequent modelling steps in order to allow one to get back to the field data, especially if adaptive mesh generation is used to improve the results or to take into account riverbed modifications or river enhancements (Secretan *et al.*, 2001).

Moreover, since any modelling activity should be submitted to proper validation procedures and fulfil minimum error criteria, it appears necessary to evaluate the DTM with respect to the quality of representation of the field data. If standardised procedures with respect to the DTM are not available, at least a simultaneous graphical representation and a visual comparison of original isolines and of those generated by the DTM can achieve this requirement (e.g. the MODELEUR software by Secretan *et al.*, 1999). Even though this activity does not guarantee the accuracy of field data sets, at least one makes sure that the DTM reproduces the terrain as measured if not as is. Distribution of numerical error can also be analysed more accurately by making use of numerical methods adapted to error estimation.

2.5 Integrating habitat value spatially –The habitat availability

In the context of determining minimum conservation flows for fish, one way to consider the river attributes consists of looking at the wetted perimeter of a limited number of sections or transects and according to the shape of these sections, finding a discharge value for which there is no significant additional gain in the wetted perimeter (Reiser *et al*. 1989). In 2D, this concept can be extrapolated to an equivalent Wetted Area ($WA(Q)$) which corresponds analytically to the integration of the flow domain with respect to its wet or dry state (Equation 5). In this approach, the riverbed surfaces only need to be wet to be considered as full habitats which, for obvious reasons related to the local habitat value ($HSI(x,y)$), may not be suitable, or effectively occupied by fish species. A minimum flow recommendation with respect to this method corresponds normally to point A on Figure 8:

$$WA(Q)=\int_D (\text{state=wet})\,dA \cong \sum_N (\text{state=wet})_i \,\Delta A_i \qquad (5)$$

where Q is the discharge and ΔA_i represents the area of a discretization element. This numerical formulation can vary according to the specific integration scheme utilised.

Similarly, habitat models aim to represent the distribution of habitats and their availability as a function of discharge in order to select the most appropriate minimum value required to maintain aquatic habitat availability to a suitable level, or to evaluate and compare future scenarios (e.g. enhancement works, climate change) with respect to historical or influenced conditions. With this approach, the wetted riverbed surfaces offer a given habitat value (HSI) that can be used as a weighting factor in the integration scheme. The habitat availability is being represented by the Weighted Usable Area (WUA) (Equation 6). This is a function of flow discharge ($WUA(Q)$) which determines the spatial distribution of the main abiotic factors controlling the local habitat suitability index (i.e. $HSI(x,y)$). It is calculated by the numerical integration of $HSI(x,y)$ over the flow domain, or a given sub-domain (e.g. a spawning ground comprised within the domain).

$$WUA(Q) = \int_D HSI(x,y)\,dA \cong \sum_N HSI_i \cdot \Delta A_i \tag{6}$$

In this expression, the area and its *HSI*(*x*,*y*) value are taken into account linearly which means that a fairly large area of poor value could provide as much habitat as a limited area of excellent value. This algebraic behaviour has raised some criticisms in the literature arguing that, in terms of biological productivity, equal amounts of *WUA*, even though they seem equivalent in quantity, will not produce the same biomass as a result. One way of avoiding this drawback of the formulae is to integrate the *HSI*(*x*,*y*) variable for values corresponding to the best habitats only (e.g. $\geq HSI_{min}(x,y)$) and neglect others (Capra *et al.* 1995).

Nevertheless, the concept of Weighted Usable Area is widely used to analyse the functional relationship between habitat availability and the flow discharge. Figure 8 illustrates this relationship. In such graphs, one can observe that typically the *WUA*s increase rapidly with discharge until a maximum habitat availability is achieved in the domain under study. With an additional increase of discharge, even though the wetted areas continue to spread out, the *WUA*s usually decline gradually, probably due to the fact that the currents become gradually too rapid, the depth too large, with corresponding increases in turbidity. However, in particular situations, this behaviour can become bi- or multi-modal, especially when more than one habitat feature occurs, each of them peaking at a different optimum flow (Leclerc *et al.*, 1996).

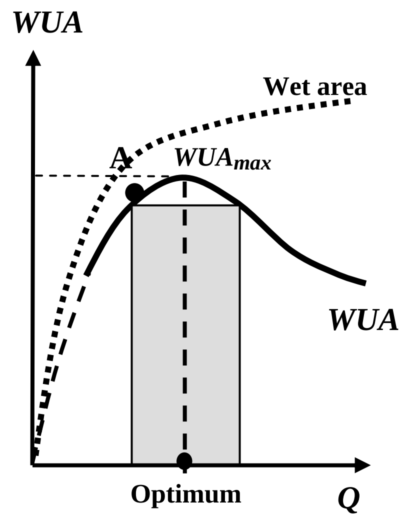

Figure 8 : The concept of wet area (*WA*) and habitat availability or weighted usable areas (*WUA)* as a function of discharge

Is the *HSI* value the only variable that can be used as the weighting factor to compute *WUA*? Some researchers (GENIVAR, 2003) proposed the use of the Probabilistic Habitat Index (*PHI*(*x*,*y*)) instead of *HSI*s to integrate the habitat availability over a flow domain. In fact, they integrate the areas where the *PHI* exceeds 0.5 in order to take into account a significant probability of presence which is greater than 50% in these zones. This means that only the best habitats are being considered in the integration scheme. This approach seems consistent with the probabilistic aspect of the *PHI* concept.

As to the fuzzy-based preference model proposed by Jorde *et al.* (2001), their suitability index is equivalent mathematically, if not conceptually, to the preference curve approaches (*HSI)* and consequently, it can also be spatially integrated over the flow domain or sub-domains in the same manner as other indices to evaluate the habitat availability *WUA*(*Q*).

2.6 Towards habitat time series

As already stated, hydrological regime generates transient flow variables which in turn can improve or deteriorate the habitat features. This aspect is not often considered explicitly in habitat studies and is open for new ideas as to how to incorporate the time factor. At least, the combined considerations of hydrological seasons and sensitive phases of the fish's life cycle provide some indications for delineating proper periods of analysis.

One way to incorporate the time factor is simply to use the hydrographs, natural and/or influenced, for building habitat time series through the *WUA*(*Q*) functional relationship (Figure 8 and Figure 9). Considering such a scheme, one can observe that the maximum amount of habitat should be achieved only during short periods of time when the optimum flow (Q_{opt}) is released from the drainage basin runoff. Any change in flow discharge from this specific point leads to a transient loss of habitat. As a corollary, any change of flow towards the optimum (increase or decrease) produces an improvement in the amount of *WUA*.

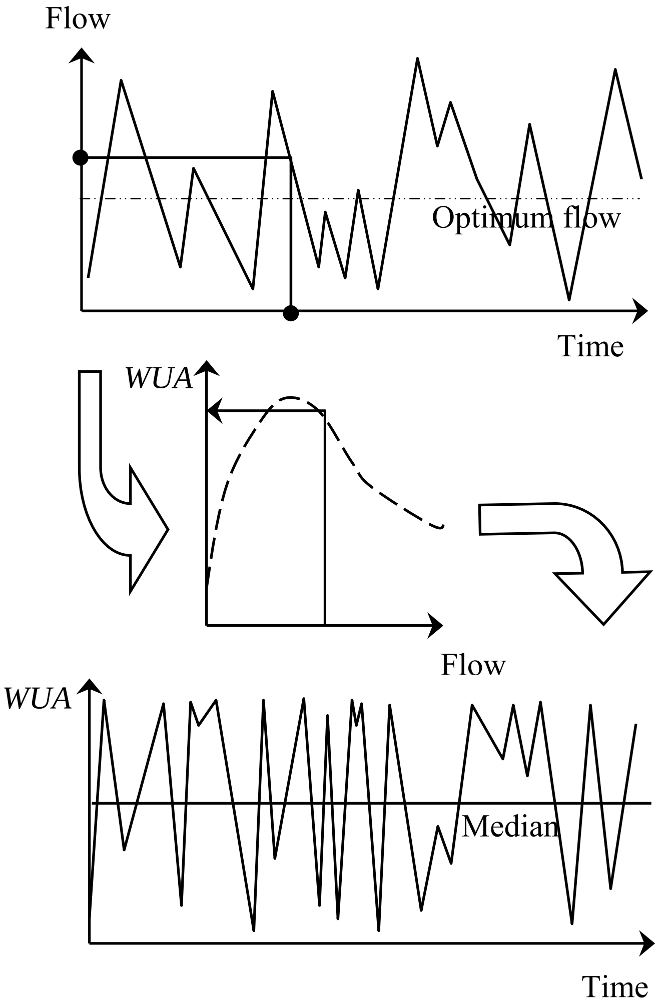

Figure 9: Time series of habitat regime as a function of hydrology

Again, some underlying hypotheses determine the interpretation of this simple scheme. Firstly, one can observe that the functional relationship between the amount of *WUA* and the discharge has two flow solutions for a single value of habitat availability. Are the two situations strictly equivalent? Probably not. For example, looking at the feeding ecology of salmonids, it is well known that the invertebrate drift load peaks during the rising limb of floods (Paraziewicz *et al*, 1996), due to the increasing velocities (shear stress), and to the escape behavior of insects which take advantage of these flow sequences to move to other locations in order to colonize new habitats.

On the contrary, during the falling limb of the flood, most of the movement is complete as insects have already settled in their new location. As a consequence, the feeding behavior is not equivalent on both sides of the flood hydrograph. Moreover, one should mention that sediment transport processes are triggered by floods above a given flow threshold, and that the concentration of particles in the water column (suspension and saltation) increases with increasing flow. As a consequence, the visibility of prey diminishes and the predators can no longer distinguish between sand and drift particles. Thus, some weighting factor based on the rate of variation of flow $Q'(t)$ could theoretically be incorporated in the model in order to take into account these transient flow dependant properties.

$$WUA(t) = \mathrm{f}(Q(t), Q'(t)) \tag{7}$$

In a context of hydro-peaking, such transient considerations could be useful to set more efficient flow recommendations.

2.7 Spatial distribution and movements of habitats

Furthermore, the change in availability of habitats during a given flood hydrograph and/or peaking flow regime is accompanied by habitat “movements”. In fact, the best habitats which were located in or close to the thalweg during low flow episodes will transfer to near the banks at higher flows while habitats in the thalweg will become less suitable because of velocities and depths that become too high compared to the preference ranges (Figure 10; Leclerc *et al*, 1994).

From an ecological point of view, a fundamental question arises : will the fish move with the shifts of the best habitats or will it stay on its “home rock” waiting for better days? In mathematical terms, this question is equivalent to considering whether the fish behaviour is Eulerian or Lagrangian. Obviously, the answer depends highly on the ecology of the fish species considered, and on their short term behaviour. For example, the main strategy of young salmonids during their feeding period is to occupy the best habitat possible (home rock) and defend it against intraspecific or interspecific competitors. The strongest fish occupy sedentarily the best spots while the weakest individuals satisfy themselves by moving around trying to access opportunistically the best areas but with limited success.

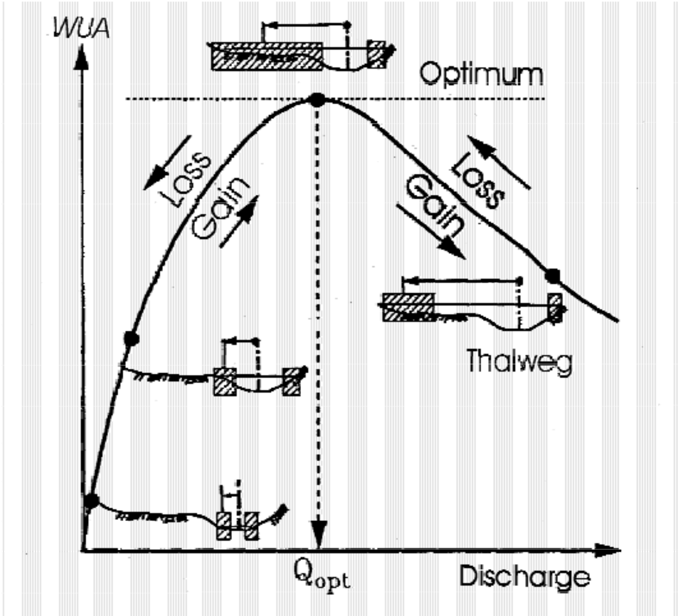

Figure 10: Habitat availability and movements with respect to flow discharge

The issue can also relate to the rate of change in flow conditions (*Q'*(*t*)). If the change takes place in a short period of time (*Q'*(*t*)>>) and leads to less favourable conditions, the fish can possibly afford to shelter in the substrate interstices and wait until better conditions come back (Eulerian strategy). Otherwise, the fish could move transiently to "better skies" (Lagrangian strategy) if the flow episodes last too long to allow sufficient feeding in the long term.

An attempt to quantify the movement of habitats was made by Leclerc *et al.* (1994). The method starts by partitioning the flow domain into sub-regions (patches) with continuous habitat distribution. Laterally, the thalweg separates the habitats on each side of the river section. Longitudinally, bed forms or flow facies play the same role. The mass centroïd of each sub-region of the habitat is then located for each reference flow event, the weighting factor being represented by the local habitat value (*HSI*). The displacements of these centroïds can be tracked in space with respect to the flow regime in order to quantify the rate of displacement (speed).

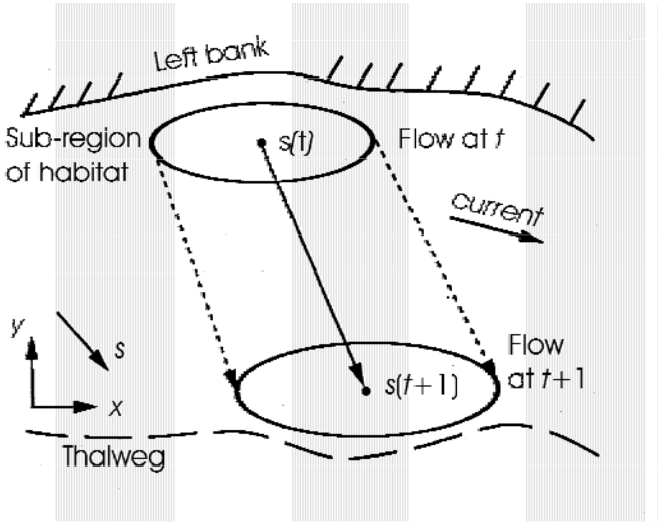

Figure 11: Displacement of sub-region of habitat with flow reduction -
Habitat feature moves to thalweg.

3. Validating habitat models in a context of uncertainties

Validation is usually recognized as an essential step in any modelling effort. As for any biological model, validation procedures constitute a great challenge and the lack of systematic and well recognised methods still limits the credibility and confidence in the approach (Caissie and El-Jabi, 2003). Model validation consists of ensuring that the choice of model parameters (mostly the biological parameters in habitat models) is adequate by comparing model results to observed values.

The most common strategy is to compare representative habitat availability values estimated by the model with the effective biomass produced within the river reach under study. This approach has produced mixed results in the past, most often non-significant. Few successes have been reported in the literature (Bovee, 1988; Jowett *et al.*, 1998; see Guay *et al.*, 2000 for additional discussion). The main reason for this limited success depends on the great deal of uncertainty characterizing the biomass variable itself which depends not only on the discharge, integrated over a long period of time (several years or the time scale of the ecosystem) but also on a wider range of secondary physical and/or physico-chemical factors (e.g. presence of deleterious contaminants). Considering that each of these factors behaves according to its own space/time scales, it becomes very difficult to pinpoint the specific value that plays the role of a limiting factor or of an explanatory variable with respect to the biomass variations. Moreover, it is often difficult to measure all those parameters in one single field survey.

In addition to abiotic factors, biomass is a very aggregative variable that is also a function of a number of independent biotic variables that belong to species population dynamics (recruitment, mortality) or interspecific relationships (competition, predation) among the aquatic communities. These processes can hardly be included in habitat models but some researchers are trying to build new frameworks dedicated to a higher synergy between the biotic and abiotic aspects of habitat modelling which also takes into account the dynamics of flow and corresponding habitat availability (Sabaton *et al.*, 2003).

A more realistic approach to validate at least the habitat preference model without the need to characterize the fish biomass has been proposed by Boudreau *et al.* (1996) and is starting to be used extensively (e.g. Guay *et al.*, 2000). Here instead, measured densities of fish are compared to local habitat values (either *HSI* or *PHI*). In this case, the emphasis is not put on total biomass, but rather on attempting to reproduce the selective behaviour of individuals for a given habitat type. This method does not provide the level of accuracy expected from a conventional validation approach, but it reduces the uncertainty associated with model parameterisation.

A great deal of uncertainty remains in defining habitat *preferenda* for a number of species and one can be interested in characterizing the typical error related to model parameterisation schemes, especially when habitat modellers have to rely on subjective expert advice only. Duel *et al.* (2003) have proposed that one should perform sensitivity analysis of habitat models to *preferendum* parameters and input variables as defined by expert advice. Monte Carlo simulations can generate large samples and provide an estimate of model uncertainties as a function of various input variables or parameters. Various authors have mentioned that only 3-4 variables play a key role and associated relative uncertainties for preference values (*HSI*) are of the order of 0-0.2, with maximum values of the order of 0.4-0.6. Hopefully, minimal uncertainties are usually found for very high (e.g. ≈1) or very low habitat values which are the most interesting from an ecological point of view.

4. Upscaling habitat models with respect to river network or catchment

The issue of upscaling is very important given the requirements of the European Water Framework Directive. CFD fits into a suite of available hydraulic modelling tools that range from high resolution, low coverage (3D CFD) to low resolution, broad coverage (1D hydraulic modelling). Most habitat model applications focus on the river reach scale which as a minimum should cover a limited number of morphological unit successions (pools, riffles, runs) representing to a certain extent the *river continuum concept*. Although this strategy provides meaningful information on the micro-habitats, such a perspective does not allow one to analyse the ecological structure and dynamics at a larger scale belonging to the river network which corresponds to the catchment scale. The classical river reach approach provides typically a "sample" of information about river habitats with respect to the entire river system but does not offer any guarantee that such results represent accurately the river network ecology.

Several distinct river reaches characterized by different proportions of geomorphic features can be modelled to enlarge the representativeness of the study. One can also develop a model covering the entire river network, or at least the main watercourse (maximum river order), such as the St. Lawrence River model (Morin *et al.,* 2003; 2000; 1997). However, the cost of such strategies often limits their applicability to large rivers with very important economic and ecological (e.g. fisheries management) issues that justify the related hydrographic characterization effort. Thus, one needs modelling strategies that allow one to up-scale (extrapolate) the reach results to the river network scale.

Several authors have addressed this question of upscaling (e.g. Duel *et al.*, 2003; Paraziewicz, 2003). Some models consider as explanatory variables average abiotic factors at the reach level without considering their local distribution (0-D; Lamouroux, 1998; Lamouroux *et al.*, 1995; 2002). These models are based on a prior aggregation of results (upscaling) obtained out of a wide sample of microhabitat models operating at a finer (1D, 2D) scale. By using average physical metrics (average reach Froude number and specific discharge, mean D_{50} diameter for substrate grain size) to describe the general abiotic conditions for a number of reaches, or even an entire hydrographic network, it is possible to obtain rough estimates of the habitat quality and availability at larger scales. These general metrics are then used to estimate habitat values for certain species or groups of similar species (guilds). It is noticeable that these parameters reflect a limnological classification of the river which considers their regime as either *lentic* which means slow currents, relatively deep, with finer sediments, or *lotic* which refers to facies with higher velocities, shallower depths and larger alluvial elements.

5. Habitat models in practical contexts

5.1 Developing new knowledge or managing water use?

Is it necessary to mention that a habitat modelling methodology, as most scientific tools, can be used either for research or for management purposes? Until now, habitat models have mostly been utilised for determining conservation flow values (see section 5.2) which correspond to management concerns. But a growing trend is observable which shows an increasing use of habitat modelling to support the development of new knowledge and integrate field information in a meaningful way, according to mass and momentum conservation laws.

In a context where major ecological study programmes are being conducted on rivers or other water bodies, distributed river flow models form the only approach allowing one to integrate several heterogeneous sets of field data, including morphology and hydrology. Empirical knowledge and field data can form very large databases on a given river, making it difficult to take into account the inherent complexity of this information. Geographical Information Systems are very useful tools to represent graphically, often in 3D, the various layers of terrain information, interpolate and combine them with simple mathematical functions. However, the algebraic capabilities of standard GIS tools remain limited and fall far short of those necessary to simulate river flows. With their predictive capabilities, steady state or transient river flow models can supply consistent and synoptic data for several abiotic variables that, otherwise cannot be economically measured (e.g. transient temperature fields, sediment transport, wave generation) simultaneously everywhere in the flow domain.

Nevertheless, if such models provide unique integrated perspectives of the flow domain, they cannot supply information for parameters not included in the conceptual framework. In fact, they cannot take into account (reductionism) every aspect of physical reality (e.g. turbulence) either, because some factors cannot yet be represented mathematically, or because they do not contribute significantly to the analysis.

5.2 Determining conservation flows – Management plans

In many countries, conservation flow practices or regulation schemes aiming to maintain aquatic ecosystems or target fish species have been applied. In some countries, such practices have been applied throughout the 20th century such as in France where a minimum of 1/40th of the mean annual flow was enforced by public regulation all the year round in river reaches influenced by dams or diversion practices (Sabaton *et al*, 2003).

Many approaches currently exist to establish water management schemes and priorities (Bérubé *et al.*, 2002; Caissie and El-Jabi 2003; King *et al.* 2002). In Canada for example, one of the guiding principles often used is the policy of no net loss of aquatic habitat, promulgated by the federal department of Fisheries and Oceans. In general, the guidelines aim to maintain sufficient flow in the river to allow for indigenous species to complete all their life stages and to maintain their population level. Ideally, these principles should also aim to maintain flow variability similar to that of the natural regime (the *physiomimesis* principle of Katopodis, 2003). Three broad categories of methods exist to achieve these objectives: *hydrological methods*, *hydraulic methods* and *habitat modelling* (Belzile *et al*. 1997). These three classes of approaches usually include (albeit to varying degrees) the notion of maintaining flow variability as determined by the natural regime and/or by anthropogenic impacts.

The simplest methods are mainly based on pure hydrological considerations (e.g., Tenant, 1976; Reiser *et al*, 1989; Russell, 1990; USFW, 1981; Bérubé *et al*, 2002; Caissie & El-Jabi, 2003) and, even though most of them consider ecohydrological seasons, they do not take into explicit consideration the instream flow needs for ecosystems or fish species.

During the late 1970s, pure hydraulic methods based on the wetted perimeter were adopted in order to offer better insights to decision makers. In addition to hydrological concerns, hydraulic methods can take into account (at least at the reach scale) the existing geomorphology and hydrodynamics in order to maintain a certain minimum wetted perimeter on given transects (Reiser *et al*. 1989; Caissie & El-Jabi 2003) or wetted area when 2D models are utilised. This approach hypothesizes that a riverbed parcel represents an available habitat as long as it is wet, which obviously overestimates the real amount of effective habitat.

Less reductionist habitat models aim to represent the distribution of suitable habitats and their total availability as a function of flow discharge. They are employed in order to select more accurately the most appropriate flow regime to maintain aquatic habitat for one (or many) aquatic species during some (or all) of their life stages. If the IFIM approach (Instream Flow Incremental Methodology, Bovee 1982) is the most commonly used method in North America and probably elsewhere in the world (Reiser *et al*. 1989), the new trend is oriented towards the use of more sophisticated 2D or even 3D discretized models of the river flow, as input data sources to habitat models.

These tools and metrics, especially the latest generation of habitat models, allow the most appropriate flow regimes for ecosystem health and conservation to be determined, in a context of multiple use of water resources. They also contribute to the rationalisation of difficult decisions and to the resolution of water allocation conflicts between conservationists and water users.

5.3 Designing impact mitigation measures or habitat enhancements

Mitigation and compensation methods accompanying habitat restoration or impact mitigation programmes may include flow management, substrate maintenance and improvement, fishways, fish stocking and engineering structures such as weirs. The latter are often presented as a method to optimize the usage of a reduced flow because they can contribute to the maintenance of proper groundwater levels, and of human water uses such as navigation; they may improve the riverine landscape by maintaining at their full extension the wetted surfaces. They may also improve habitat availability for fish recruitment and overwintering (Britain, 2003).

Weirs can ensure a more constant wetted perimeter in upstream reaches and may help to concentrate flow, which can provide improved conditions under reduced discharge. However, Britain (2003) mentions that the conversion of a river system to a more lentic environment may lead to an increase in macrophyte production and may also favour modifications in the fish community structure (e.g. from a salmonid-dominated river to a cyprinid one). These structures may also be seen as obstacles to fish migration and increase fine sediment deposition. Some aquatic birds can benefit from such modifications, probably due to the increase in benthic productivity.

Weirs can be made of wood, concrete or rocks, according to the stated objectives, prevailing environmental and hydraulic conditions, as well as available material. A comprehensive follow-up program is advised upon implementation of such structures, in order to optimize flow management schemes or improve structure design when possible (Britain, 2003).

These intrusive methods do not traditionally rely on prior habitat modelling to justify their construction. The usual design criteria are often limited to the estimation of the wetted perimeter and the associated flow patterns. Weirs will often modify significantly the habitat structure from lotic to lentic, with lower velocities and higher sedimentation after construction. A coupling of the design process with habitat models is therefore a desirable objective because it may help to maximize the outcome of such structures on aquatic habitats. Traditional PHABSIM models may not be the ultimate tool for such coupling. Distributed 2-D models offer the flexibility required to test the effectiveness of various structure designs.

The recent development of more sophisticated structures, such as adjustable sills, allows for an adaptive varying geometry and may permit a better modulation of the flow patterns. A periodic increase in wetted perimeter can be synchronized with spawning or migration requirements of certain fish species. These concepts appear feasible at reasonable costs with current technology and know-how.

6. References

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