

A critique of IFIM—instream habitat simulation in the New Zealand context

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Henry R. Hudson, Andrea E. Byrom, W. Lindsay Chadderton

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CONTENTS

Abstract	5
<hr/>	
1. Introduction	6
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2. Flow objectives and terminology	7
2.1 Minimum flow	7
2.2 Instream flow	8
2.3 Environmental flow regimes	9
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3. IFIM-physical habitat simulation	10
3.1 IFIM framework and applicability	10
3.2 PHABSIM compared with RHYHABSIM	12
3.3 Physical habitat simulation overview	13
<hr/>	
4. Describing rivers	15
4.1 Study area boundaries	15
4.2 Stream segments	16
4.3 Representative reaches	17
4.4 Mesohabitat typing = habitat mapping	18
4.5 Critical reaches	20
4.6 Transect location	21
4.7 Microhabitat suitability	23
4.7.1 Depth and velocity	23
4.7.2 Substrate	24
4.7.3 Cover	26
4.8 Hydraulic-habitat simulation	26
<hr/>	
5. Habitat suitability criteria (HSC)	28
5.1 Applicability and transferability	28
5.2 Sampling problems	29
5.3 Quantifying habitat suitability	31
5.3.1 HSC development	31
5.3.2 WUA computation	32
5.3.3 Biological meaning of WUA	32
<hr/>	
6. Additional considerations in physical habitat simulations	35
6.1 Biotic factors	36
6.1.1 Competition and predation	36
6.1.2 Fish migrations including diadromy	37
6.2 Abiotic factors	37
6.2.1 Water quality	38
6.2.2 Temperature	38
6.2.3 Refugia/cover	39
6.2.4 Groundwater linkages	39
6.2.5 Instream flow variability	39
6.2.6 Lateral linkages	41

6.2.7	Longitudinal linkages	42
6.2.8	Channel evolution	42
6.3	Microhabitat interactions	43
7.	Verification and validation	44
8.	Training	45
9.	Resource Management Act considerations	45
10.	Summary and recommendations	47
10.1	Problems describing rivers	48
10.2	Problems with habitat suitability curves	49
10.3	Problems with biological meaning of WUA	50
10.4	From minimum flows to environmental flows	51
10.5	Suggested approach	51
11.	Acknowledgements	52
12.	References	53
<hr/>		
Appendix 1		
<hr/>		
	Common and scientific names of animals mentioned	69

A critique of IFIM—instream habitat simulation in the New Zealand context

Henry R. Hudson¹, Andrea E. Byrom², W. Lindsay Chadderton³

¹ Environmental Management Associates Ltd., Christchurch hudsonh@es.co.nz

² Landcare Research, PO Box 69, Lincoln

³ Department of Conservation, PO Box 112, Hamilton

ABSTRACT

The need to sustain the ecological values of rivers is widely recognised and embraced in policy and legislation. Here we examine a widely used approach for evaluating effects of changing flow regimes, the Instream Flow Incremental Methodology (IFIM), in particular physical habitat simulation using PHABSIM/RHYHABSIM. We review IFIM procedures and discuss limitations of habitat simulation with specific reference to New Zealand. Our objectives are to encourage a critical re-evaluation of IFIM and improve its application so that it takes into account known but frequently unaddressed problems. These generic problems are compounded by a limited knowledge of many species, the lack of rigorous description of habitat requirements, the fact that habitat suitability curves have been developed for a very limited range of conditions and a narrow view of flow requirements. The New Zealand IFIM experience has usually been limited to an evaluation of the effects of minimum streamflows on various life stages of a few species of fish and on food production for a small reach of stream. Rarely has consideration been given to transferability of results, or changes in water temperature or water quality with changing flow regimes. River mouth openings, flushing flow requirements, maintenance of lateral and longitudinal stream processes and maintenance of river channel processes are essential components of an environmental flow assessment, but have often not been considered. We also found that the IFIM process is often confused with one of its basic steps—using models to simulate a relationship between streamflow and in-channel physical habitat.

Keywords: IFIM, PHABSIM, RHYHABSIM, instream flow, environmental flow, limitations, critique

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1. Introduction

Water engineering has profoundly improved living standards by: providing fresh drinking water, water for irrigation, energy and transport and flood control—but often at significant environmental, and human, cost (Revenga et al. 2000; Mueller & Marsh 2002). The need to sustain the ecological values of rivers is widely recognised and embraced in policy and legislation (e.g. the Canadian policy of No-Net-Loss of productive habitat: DFO 1986; New Zealand Resource Management Act 1991: MFE 1998; the Australian policy for water resources: Cullen 1994; South African Water Law 1998: King et al. 1999). However, establishment of flow requirements to sustain these ecological values of rivers is one of the most contentious issues facing water managers worldwide (Dunbar & Acreman 2001).

Here we examine a widely used approach (Dunbar et al. 1998; King et al. 1999) for evaluating effects of changing flow regimes, the **Instream Flow Incremental Methodology (IFIM)** (Bovee 1982). There has often been confusion between the IFIM process and the tools used and steps taken (Table 1) to quantify changes (Stalnaker et al. 1995) (e.g. MFE 1998). IFIM addresses the decision making environment as well as the techniques for quantifying incremental differences in habitat in a stream that result from alternative flow regimes (MESC 2001).

We have few concerns about IFIM as a process. Our concerns relate to one of the key IFIM tools—physical habitat simulation using PHABSIM¹ (Milhous et al. 1981, 1989; MESC 2001) (or RHYHABSIM, the simplified variant used in New Zealand: Jowett 1996). In New Zealand, physical habitat simulation of flows within the river channel has been the focus, with little discussion of problems or uncertainties being articulated in the literature (but see Scott & Shirvell 1987; Castleberry et al. 1996). For example, in the ‘Flow guidelines for instream values’ the MFE (1998) stated IFIM ‘... has been used in a number of situations in New Zealand and is well suited to the physical and ecological characteristics of New Zealand rivers.’ In contrast, the appropriateness of PHABSIM has been questioned internationally and alternative methodologies developed (e.g. Australia: Arthington et al. 1992, Thoms & Swirepik 1998; China: Xia et al. 2001; Germany: Freistühler et al. 2001; Italy: Buffagni 2001; South Africa: King & Louw 1998, Brown & King 2000). According to Day & Hudson (2001) and Hudson (2002) a broader perspective is being increasingly adopted.

In North America, where IFIM-PHABSIM is widely used, major differences of opinion concerning habitat changes and biological responses to changing flow regimes have occurred in expert testimony (e.g. Colorado: Gordon 1995; California: Williams et al. 1997). An expert panel convened through the court considered uncertainty and instream flow standards (Castleberry et al. 1996; Williams et al. 1997). All participants agreed that currently no scientifically defensible method exists for defining the instream flows needed to protect particular species of fish or aquatic ecosystems. However, the panel was split

¹ Physical HABitat SIMulation (PHABSIM); River HYdraulics and HABitat SIMulation (RHYHABSIM).

TABLE 1. IFIM PROJECT PHASES AND ACTIVITY SUMMARY (BASED ON STALNAKER ET AL. 1995 AND MESC 2001).

Step	Activity
1 Problem Identification Institutional Analysis Physical Analysis	Identify/contact legitimately interested parties. Determine: <ul style="list-style-type: none"> • Geographic extent • Possible factors limiting success.
2 Study Planning Strategy Design Technical Scoping	Define: <ul style="list-style-type: none"> • Temporal and spatial scale • Important variables (e.g. water quality, habitat availability) • Plan data collection and analysis. Describe: <ul style="list-style-type: none"> • Hydrologic time series and biological reference conditions (historic records or simulated) • Historic critical events (e.g. droughts, spills). Identify: <ul style="list-style-type: none"> • Critical reaches (e.g. impediments to migration) • River segment types (e.g. braided, meandering) • Representative reaches. Select: <ul style="list-style-type: none"> • Target species or guild • Select or develop habitat suitability criteria
3 Study Implementation Data collection Model calibration Simulation Habitat Models Interpretation	Measure within representative and critical reaches: <ul style="list-style-type: none"> • Velocity, depth, substrate and cover • Temperature, pH, dissolved oxygen, biological parameters ... • Verify model assumptions with site data • Calibrate models with site data • Describe relation between stream flow and stream habitat utility • Determine baseline habitat time • Evaluate water quality and temperature effects • Develop weighted usable area (WUA) vs. discharge function (PHABSIM) • Generate time series of daily or monthly WUA. Determine habitat bottlenecks
4 Analysis of Alternatives	Test alternatives <ul style="list-style-type: none"> • Factor in physical and institutional constraints on water management • Compare scenarios with baseline habitat time series.
5 Problem Resolution	Negotiate multiple-use, water budget approach <ul style="list-style-type: none"> • Trade-off habitat cost/benefits, feasibility, risk and economics • Agreement → implement flow regime • No agreement → RE-START
6 Verification and Validation	Use adaptive management approach <ul style="list-style-type: none"> • Post-project monitoring and evaluation • Reassess flow objectives/flow regime

on the future role of physical habitat simulation as undertaken in IFIM. Two views emerged:

1. With modification and careful use, IFIM-habitat simulation might produce useful information; and
2. IFIM-habitat simulation should be abandoned.

There was agreement that users of PHABSIM, or some modification of it, must take into account the following (Castleberry et al. 1996):

- Sampling and measurement problems associated with representing a river reach with selected transects and with the hydraulic and substrate data collected at transects.
- Sampling and measurement problems associated with developing the [habitat] suitability curves.
- Problems with assigning biological meaning to the habitat statistic estimated by PHABSIM, weighted usable area (see below).

In addition to methodological, morphological and ecological problems associated with IFIM, Bovee et al. (1998) (the originators of IFIM) stated: 'IFIM ... is widely misconstrued, misinterpreted, and in some cases misused'.

In this paper we briefly review flow regime terminology and IFIM procedures, and then discuss limitations of habitat simulation—with specific reference to New Zealand conditions, and how habitat simulations have been applied here. Our objectives are to encourage a critical re-evaluation of IFIM and improve the way in which IFIM is used.

2. Flow objectives and terminology

There are inconsistencies in the definitions and usage of flow objective terminology. Dunbar & Acreman (2001) attributed the origin of the term 'instream flow' to North America, while 'environmental flow' has been commonly used in South Africa and Australia. They use the terms interchangeably to describe '... the use of freshwater science and hydrology to manage the ecological impact of river regulation'. In fact, 'minimum flow', 'instream flow'/'in-channel flow' and 'environmental flow' are not synonymous—the terminology reflects a paradigm shift (MESC 2001; Hudson 2002).

2.1 MINIMUM FLOW

Minimum flows are often a subjectively determined water level or flow, retained for the purpose of survival of a particular fish species. As discussed in Stalnaker (1994) and Stalnaker et al. (1995), in the western United States the concept of retaining 'minimum flows' in streams developed in response to the severe declines in fisheries and amenity values because of water developments.

This exploitation of water for 'off-stream' benefits, such as irrigation agriculture, with little regard to the degradation of the instream aquatic environment, was the norm in water resource management internationally (e.g. Collier et al. 1996). Water flowing to the sea was considered wasted: 'The residual flow (at the estuary) should be zero ... unless there are reasoned arguments to the contrary' (Law 1972; quoted in Dunbar et al. 1998). The reasoned arguments included allowing some flow for fish passage for sea-run trout² and salmon and for cooling holding pools to prevent fish kills because of high temperatures.

Instream flow assessment methods were developed in the 1960s and 1970s based on hydrologic analysis of the water supply and hydraulic geometry relations, coupled with observations of habitat quality and fish ecology. Application of these methods usually resulted in the specification of a minimum flow below which water could not be withdrawn or withheld, usually to ensure the survival of a target species of fish (often salmonids) in a river.

One of the serious problems with this 'low flow' approach was that biologists distinguished between only two river conditions with respect to fish habitat: the level below which disaster would occur and all other flows (MESC 2001). While aquatic life could tolerate an extreme low flow event over the short term, over extended periods these flows were unsustainable.

2.2 INSTREAM FLOW

Instream flows are an objective balance of the flow regime needs of in-channel users (e.g. fish and water sports) and off-channel uses (e.g. irrigation). In the early 1970s a series of reviews and workshops concluded: 'In general ... it was not possible to systematically relate instream habitat values to instream flows using the technologies existing at that time' (Nestler et al. 1989). This led to the development of the Instream Flow Incremental Methodology (IFIM) by a multi-agency group led by the United States Fish and Wildlife Service.

'IFIM is based on the analysis of habitat for stream-dwelling organisms under alternative management treatments. One could logically question why habitat was chosen as the decision variable ... when there are so many other factors (such as stream productivity or fishing mortality) that can potentially influence fish populations. The simplest reason ... is that IFIM was designed to quantify environmental impacts, and impacts [on] ... habitat are the most direct and quantifiable' (Stalnaker et al. 1995).

PHABSIM (Milhous et al. 1981, and its variants) predicts how physical habitat (depth, velocity, substrate and sometimes an index of cover) changes with flow and combines this information with habitat suitability criteria (HSC) to determine an index of the amount of habitat available over the range of streamflows (the weighted usable area: WUA).

Management alternatives are evaluated for various uses (e.g. hydro-peaking flows, recreation, fisheries and downstream consumption) by incrementally

² Scientific names are listed in Appendix 1.

changing the flow. The outcomes of the simulations and flow negotiations are usually recommendations for a range of flows (e.g. seasonal, wet and dry year flows) necessary for fish passage, to provide sufficient instream habitat for particular species and life stages and to ensure that flow-dependent water quality requirements are met (e.g. temperature and dissolved oxygen, which are modelled with other tools) (Stalnaker et al. 1995). In addition, instream flows may also be specified to remove excessive fine sediment from the riverbed (flushing flows, Milhous 1996).

Theoretically, IFIM-PHABSIM applications are not limited to low flows in a fixed channel but, in practice, dealing with channel evolution is problematic (Bovee et al. 1998), and as of 2001 IFIM programs have lacked guidelines or models for assessing out-of-channel flow requirements (MESC 2001). Usually IFIM-PHABSIM use has been limited to determining low flow requirements within existing stream channels (so that instream flows are synonymous with in-channel flows), for a fixed bed, for selected river reaches.

2.3 ENVIRONMENTAL FLOW REGIMES

Environmental flows address ecosystem needs, and maintenance of flow-dependent ecosystem structures and processes, at various scales (e.g. Thoms & Parsons 2002). Environmental flows provide a flow regime for the river corridor (i.e. the channel, the floodplain and the transitional upland fringe) and receiving waters (e.g. lake, coastal zone), for the purpose of maintaining ecosystem structure (e.g. wetlands, oxbow lakes) and processes (e.g. nutrient cycling; sediment flux) (Hudson 2002).

In the river corridor, flow regimes are required to maintain lateral (riverine-riparian-floodplain), longitudinal (headwater-riverine-estuary) and vertical (riverine-groundwater) processes (e.g. nutrient dynamics and energy flow) (Junk et al. 1989; Ward & Stanford 1995). Hill et al. (1991) have argued that multiple flow regimes are needed to maintain biotic and abiotic resources: (1) floodflows, that form floodplain and valley features; (2) overbank flows, that maintain surrounding riparian habitats, adjacent upland habitats, water tables and soil saturation zones; (3) in-channel flows that keep immediate stream banks and channels functioning; and (4) in-channel flows that meet critical fish requirements. Further, there is a need to determine how altered streamflows affect channels, transport sediments and influence vegetation (Hill et al. 1991). Indeed, various studies have shown that floodplain sediment is replenished by overbank flows and sedimentation and that it is crucial for the germination of cottonwood trees (Rood & Mahoney 1993); organic carbon budgets (McGinness et al. 2002; Olley 2002); and fish habitat (Snyder et al. in press) (see Section 6.2.6).

For environmental flows, downstream effects must be explicitly considered. Flow manipulations can modify water quality in deltas, estuaries and adjacent wetlands (e.g. salt wedge position, California Water Commission 1994; Abam 2001); can limit fish passage (e.g. river mouth closure, McDowall 1995); and habitat availability (e.g. Oyebande, 2001); and can significantly modify productivity (Yin et al. 1997), morphology and hydrodynamics of the coastal zone (Kirk 1991; Abam 2001).

The challenge is to determine the critical features of the frequency, duration, rate of change and timing of flows to be applied to particular rivers (e.g. Richter et al. 1996, 1997; Poff et al. 1997) so that the ecosystem needs, as determined by comparison to reference sites or conditions, will be met (Hudson 2002).

Approaches to determining the environmental flow requirements have been developed in Australia (holistic approach: Arlington & Zalucki 1998, Arthington et al. 1992; expert panel assessment method: Swales & Harris 1995; scientific panel assessment method: Thoms et al. 1996; Thoms & Swirepik 1998); South Africa (building block methodology: Arthington 1998, King & Louw 1998; DRIFT: Brown & King 2000); and North America (Bartholow et al. 1993; Nestler et al. 1993; Stanford et al. 1996; Toth 1996; Richter et al. 1996).

3. IFIM—physical habitat simulation

3.1 IFIM FRAMEWORK AND APPLICABILITY

IFIM is a decision-making tool that includes quantifying the incremental differences in instream habitat that result from alternative instream flow regimes. The purpose of physical habitat simulation is to relate changes in streamflow to changes in physical habitat for various life stages of a species of fish or other organisms, for food production, for riparian vegetation or for a recreational activity. Changes in streamflow may be linked, through biological considerations, to environmental and social, political and economic outcomes (Stalnaker et al. 1995).

As noted, the simplest reason for basing the analysis on habitat is that impacts on habitat are the most direct and quantifiable (Stalnaker et al. 1995). Further, limits of habitat supply must, by some means, control the size and dynamics of fish populations (Nehring & Andersen 1993; Minns et al. 1995; Cunjak & Therrien 1997).

Two of the most important aspects of any PHABSIM study are that it is set in the IFIM framework (Table 1) and that it is adequately planned (MESC 2001). There are several phases and various activities in an IFIM analysis and the first, problem identification, determines the scope of the problem and determines if PHABSIM is appropriate and sufficient.

‘In any IFIM application, care should be taken to ensure that the factors that control the instream habitat have been fully considered prior to the use of PHABSIM. For example, ... where the sole limit to habitat is water quality [an] ... IFIM study in such a situation may be appropriately limited to only water quality induced limits to habitat quality and quantity. That is, an IFIM study may be appropriately conducted without use of PHABSIM if physical habitat is determined not to be a major limiting variable ... Use of PHABSIM for a study and interpretation of WUA results will be different if physical habitat is one of the factors limiting target species populations than if it is not limiting’ (MESC 2001).

'The IFIM study may also incorporate models and/or expert knowledge to assess water quality [e.g. QUAL-2E: Brown & Barnwell 1987], water temperature [e.g. see Bartholow 2002], geomorphology, or other characteristic features of the stream and its current and proposed flow regimes that could influence habitat or populations of aquatic organisms within the stream corridor.' (MESC 2001). These variables must be evaluated on a stream system basis (Stalnaker et al. 1995).

For comprehensive reviews of the IFIM process and guidance consult Stalnaker et al. (1995); Bovee et al. (1998) and MESC (2001).

The New Zealand instream flow guidelines (MFE 1998) provide an IFIM-type framework, but describe IFIM as a habitat assessment method, with explicit reference to RHYHABSIM. The emphasis in New Zealand IFIM studies has been on habitat modelling per se, as undertaken with the original PHABSIM models (Milhous et al. 1981).

3.2 PHABSIM COMPARED WITH RHYHABSIM

PHABSIM and RHYHABSIM both calculate water depths and velocities (there are fewer options in RHYHABSIM), and both use HSC to compute an index of the amount of microhabitat available for different aquatic species or uses, at different streamflows. However, PHABSIM has evolved to include additional capabilities such as nose velocities, point velocities, shear stress, conditional cover, conditional velocity, contiguous width, minimum area exclusion, proximity to shore habitat, stranding, various aggregation methods and competition analysis (MESC 2001).

These developments in PHABSIM capabilities are illustrated for conditional velocity. Drift-feeding fish are known to feed across velocity differentials, from a slow-moderate focal point into faster surrounding water (Hayes et al. in press). PHABSIM can calculate habitat area conditioned by adjacent velocities within a specified distance (e.g. the sight feeding distance). Usable area is calculated as the sum of lower velocity habitat where there is higher velocity feeding habitat within the specified search distance (MESC 2001).

In addition, a new generation of habitat management models, for use within the IFIM framework, have been developed. Modeling involves integration of contemporary fish population models with spatio-temporal habitat models. These models require considerable knowledge of the fish population, including seasonal and annual mortality rates, seasonal patterns of movement within the stream network and estimates of habitat carrying capacity for each life stage (e.g. SALMOD: Bartholow et al. 1993, 1997; Williamson et al. 1993; e.g. SIAM: Bartholow et al. 2003).

In this critique we limit discussion primarily to the more basic functions of PHABSIM in the IFIM process.

3.3 PHYSICAL HABITAT SIMULATION OVERVIEW

PHABSIM/RHYHABSIM simulation is undertaken in five steps.

1. Study areas are selected.
2. The hydro-geomorphology of the study area is surveyed and hydraulic models calibrated so that changes in depth and velocity can be simulated at different streamflows.
3. HSC are selected or developed to represent how 'suitability' for a species and life stage varies with habitat variables (specifically depth, velocity, substrate, and perhaps cover).
4. The hydraulic model is coupled with the HSC to simulate how WUA (the index of habitat quality-quantity), varies with streamflow.
5. WUA-streamflow relationships for individual species and life stages are calculated. These relations are interpreted to develop an instream flow recommendation.

Study reaches are delineated and study sites are selected based on an evaluation of streamflow changes downstream (tributaries and diversions); spatial zonation of channel form, fish species and water quality; and identification of critical habitat types, representative reaches or habitat units (Section 4).

One-dimensional (1D) hydraulic modelling, as undertaken by PHABSIM and RHYHABSIM, is based on measurements of water depths and water velocities at specific locations across a stream at different flows (three flows are recommended: MESC 2001). Depending on the modelling option selected in PHABSIM, water surface elevations may also be measured. These data are used to calibrate the hydraulic models and then predict depths and velocities across the channel at flows different from those measured (Stalnaker et al. 1995).

Numerous points are measured across the channel (often 20 or more). Each point forms a cell with a width determined by the measurement spacing across the channel, and the length by the sampling strategy (e.g. half the distance to the downstream transect, or a proportion of the total length of channel represented by the sampling location). These 'stream cells' are typically metres wide and tens to hundreds of metres long.

Some recent studies have replaced transect-based 1D hydraulic models with two-dimensional (2D) and three-dimensional (3D) models that may allow better definition of depths and velocities in the modelled reach (Crowder & Diplas 2000; Kondolf et al. 2000; Waddle et al. 2000). These more sophisticated models require large amounts of data for their detailed descriptions of the channel geometry, with the accuracy of the results dependent on the accuracy and spatial resolution of the topographic measurements (e.g. Leclerc et al. 1995; Ghanem et al. 1996). Models may have fixed cell sizes (e.g. Rangitata River: 2 m by 2 m, Duncan & Hicks 2001), or may define an irregular triangulated mesh where the survey intensity increases with the increasing complexity of the bed. An example of the later is River2D (Blackburn & Steffler 2002), which evolved from a combination of PHABSIM with CDC2D (Ghanem et al. 1996), and has been used by the originators of IFIM (Waddle et al. 2000). River2D maps substrate to define bed roughness not only for hydraulic modelling but also as an attribute for habitat modelling (Blackburn & Steffler

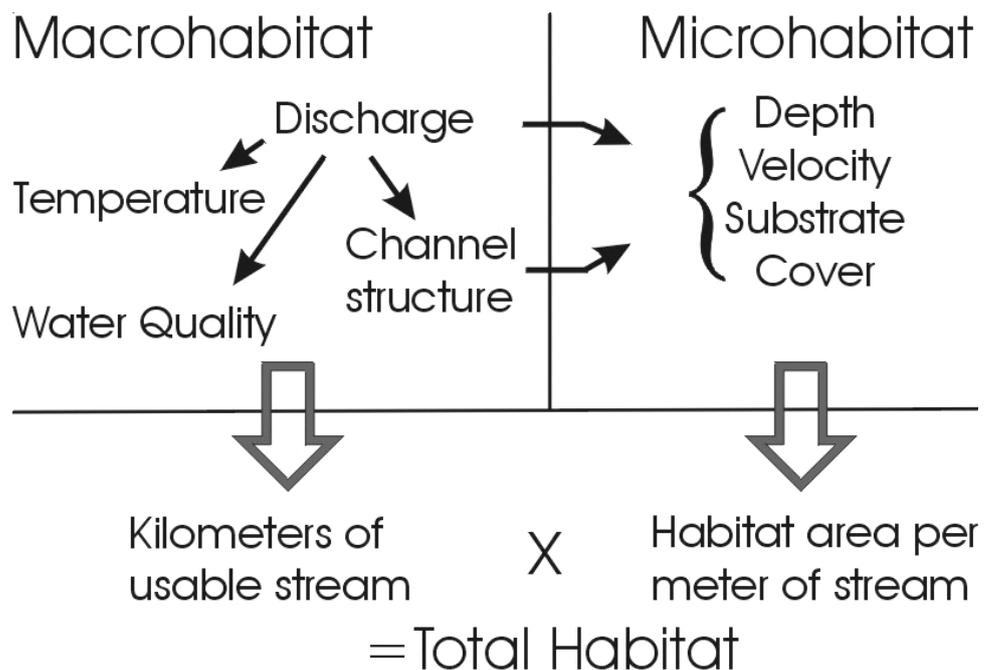
2002). Such 2D models use ‘shallow water equations’ to predict the variation in water depth and velocity across the channel and downstream.

In the next step, habitat availability is simulated for each cell using indices that assign a relative value between 0 and 1 for each habitat attribute, to indicate how suitable that attribute is for a particular species and life stage. The univariate curves that are generated are the HSC or preference curves, and may reflect habitat preferences or suitability as defined by water depth, velocity and channel index (channel index represents substrate (in RHYHABSIM) and/or cover or other immobile variables important in determining the physical habitat requirements of the target species; MESC 2001). The HSC may be derived from existing literature, expert opinion or data collected (such as by snorkelling and electro-fishing) (Bullock & Gustard 1992; Gordon et al. 1992; Jowett & Richardson 1995).

‘PHABSIM results are very sensitive to HSC. A PHABSIM analysis report should either justify transferring habitat suitability criteria developed elsewhere to the study stream, refer to development of HSC curves specifically for the study in question, or document the agreements by which consensus on HSC curves for the study was attained’ (MESC 2001). Various approaches are taken to factor assorted biases out of the habitat suitability data, but HSC remain indices that are used as weights of suitability (Stalnaker et al. 1995). Calibration, verification and validation of short-term habitat use, individual fish behaviour, long-term habitat availability and population response are integral to IFIM but, up to 1995, have seldom been undertaken (Stalnaker 1994; Stalnaker et al. 1995).

Estimates of depth, velocity and substrate at different flow levels are combined with the HSC to provide an index of habitat suitability for individual stream cells of the streambed. The weighted values for all cells are summed to produce WUA. There are various ways to sum these data in PHABSIM (but not in RHYHABSIM). Total habitat is synthesised in most studies by integrating large-

Figure 1. Total habitat is calculated by combining microhabitat (WUA) with other macrohabitat factors (after Stalnaker et al. 1995).



scale macrohabitat variables with small-scale microhabitat variables (Fig. 1), rather than keeping the variables separate.

Results from PHABSIM can be used alone or in a suite of habitat time series programs to generate monthly or daily habitat time series from the WUA-streamflow relationship and time series data on streamflow. The baseline habitat time series estimates how much habitat in total would be available for each life stage of each species over time (Stalnaker et al. 1995). Another more dynamic approach, SALMOD—a life history-based fluvial salmonid population dynamics model, has been developed in which a temporally variable habitat supply (i.e. WUA) is linked with time (flow hydrograph) (Bartholow et al. 1993).

4. Describing rivers

‘In order for a simulation model to be useful in the planning process, its output must be capable of extrapolation into space and into time. ... The study area is a sample of the conditions found in a larger stream reach, which allows the extension of the results obtained from the study area over the larger reach’ (Bovee & Milhous 1978).

IFIM handbooks place considerable emphasis on delineating study area boundaries and river segments (long sections of river with similar water quality, flow and morphology), on selecting representative reaches or habitats (short sections of river, or habitats, that represent each segment) and placing transects (survey lines across streams) for microhabitat description (Bovee 1997; Bovee & Milhous 1978).

4.1 STUDY AREA BOUNDARIES

During the problem identification phase of an IFIM analysis, the spatial and temporal boundaries must be determined.

- For evaluation of water abstraction impacts, the upper study area boundary could be taken as the uppermost point of abstraction in the stream of interest, or the point where upstream migration of target fish species is prevented (e.g. a dam or waterfall). The downstream study area boundary should be placed where the effects of the proposed action is no longer detectable (Bovee et al. 1998). In practice the lower boundary has often been located where the stream converges with a large reservoir, another river or the ocean (Bovee et al. 1998).
- The baseline for comparison of ‘before’ and ‘after’ must be determined (Bovee & Milhous 1978). MESC (2001) cautions that realistic comparisons must be made. For abstractions or impoundments, the historic natural flow regime has often been used as a baseline (i.e. flows without abstraction or impoundment). For example, for the Rangitata Water Conservation Order hearings,

alternative flow regimes were compared with the habitat availability for the natural 7-day mean annual low flow (Hudson 2001c).

4.2 STREAM SEGMENTS

'The stream segment is the basic habitat accounting unit of the IFIM, a first order subdivision of the study area. Stream segments are relatively long sections of stream, typified by a geographically homogeneous flow regime. The discharge at the top of a segment is about the same as at the bottom ($\pm 10\%$ or so). The overall channel geomorphology (slope, sinuosity, channel pattern and structure, geology, and land use) is usually consistent within segment boundaries' (Bovee 1997).

As stated in Section 3.3, total habitat is calculated by integrating large-scale macrohabitat variables (the stream segment characteristics) with small-scale microhabitat variables collected at the reach scale (see Section 4.3). The product of the length of useable stream (macrohabitat, in metres) and habitat area per unit length of stream (WUA in m^2/m) is the total useable habitat (Fig. 1).

The necessity for delineating stream segments of like character is obvious—significant changes in channel and flow characteristics, water quality, flora and fauna generally occur as rivers flow downstream from the headwaters to the sea. For example, in the Tongariro River, New Zealand, there were large differences in the optimum flow for rainbow trout adults between the lower river boulder-dominated reach ($50 \text{ m}^3/\text{s}$) and downstream sand bed reach ($20 \text{ m}^3/\text{s}$) (Hudson 2000). This longitudinal variation may also be dynamic in time, often with greater heterogeneity at lower flows (e.g. Heggenes 1996; Stalnaker et al. 1996; Giberson & Caissie 1998; Hilderbrand et al. 1999).

Internationally, differences in channel character have been found to affect sensitivity to changing streamflows (Bovee & Milhous 1978). The effect of mesohabitat has been widely recognised (e.g. Beschta & Platts 1986; Heede & Rinne 1990; Rabeni & Jacobson 1993) and has been associated with up to a five-fold difference in mean WUA for Atlantic salmon fry and a three-fold difference for parr between reaches with different channel form on the same river at the same discharges (Payne & Lapointe 1997). In the Rangitata River, optimum flows for food production and brown trout adults occurred at greatly different flows between (and within) channel types (single thread and braided) for a river with similar slope and discharge over the study reach (Hudson 2001c) (Table 2).

Examples of downstream changes in species abundance and composition can be found in Otago, where several species are generally found near the coast (e.g. banded kokopu and inanga), while others are more widespread (Allibone 1997). McDowall (1993) discusses possible reasons for these types of patterns.

In our experience, IFIM studies in New Zealand have usually neglected an initial rigorous delineation of river segments. This is considered to lead to difficulties in extrapolating reach results with any degree of certainty to larger stretches of river (Bovee 1982; Kershner & Snider 1992; Rabeni & Jacobson 1993; Maddock & Bird 1996).

TABLE 2. EFFECTS OF CHANNEL TYPE ON HABITAT AVAILABILITY WITH CONSTANT GRADIENT AND DISCHARGE FOR THE RANGITATA RIVER. (Remodelled with hydraulic data from Jowett 1998 and Duncan & Hicks 2001, based on Hudson 2001).

REACH	RIVER (km)	CHANNEL TYPE AND DOMINANT SUBSTRATE	OPTIMUM FLOW (m ³ /s) & WUA (m ² /m)			
			FOOD PROD.		BROWN TROUT	
Peel forest	48	Single channel, large boulders	10 m ³ /s	18 m ² /m	>50 m ³ /s	>7 m ² /m
Arundel	36	Semi-braided, boulders	>80 m ³ /s	71 m ² /m	50–65 m ³ /s	19 m ² /m
Arundel bridge	33	Single channel, boulders	35 m ³ /s	28 m ² /m	10 m ³ /s	12 m ² /m
Ealing	10	Braided, gravel-cobble	>80 m ³ /s	76 m ² /m	25 m ³ /s	16 m ² /m

Jowett (1998) modelled flows <50 m³/s; Duncan & Hicks (2001) <80 m³/s

4.3 REPRESENTATIVE REACHES

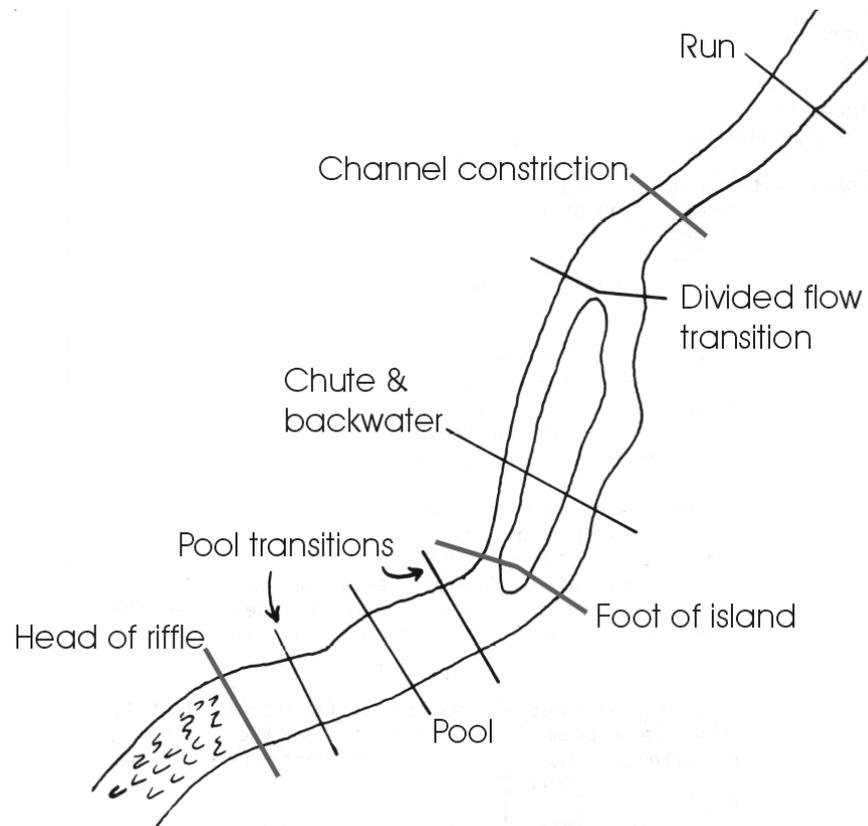
River or stream segments are sub-divided into short reaches in which detailed microhabitat measurements are undertaken. The underlying premise of the representative reach survey is that mesohabitat types (e.g. pools, riffles, runs) tend to occur in a somewhat repetitive pattern. These patterns are well documented, and well known, in alluvial channels (e.g. riffles tend to be about 7–10 channel widths apart) (e.g. FISRWG 1998). In some situations (e.g. bedrock-controlled channels) these repetitive patterns do not necessarily occur, and alternative sampling strategies are required (e.g. habitat mapping; Morhardt et al. 1983).

The representative reach is commonly 10–15 channel widths long (i.e. two morphological cycles), and is assumed to contain essentially all of the mesohabitat types, in the same proportions, as the stream segment (Bovee 1997). Various approaches can be used to select a reach to represent the stream segment (Bovee 1982) and Dolloff et al. (1997) have shown that the method of selecting representative reaches is very important. In terms of characterising the microhabitat attributes of the reach for 1D modelling, transects are usually placed at the hydraulic controls and major habitat features in the reach (Fig. 2). For 2D modelling, aerial representations of the stream are used to select appropriate river reaches.

We found that surveyed reaches have often been very short in New Zealand IFIM studies (a few channel widths in length) and often did not include the range of habitats in the segment. This practice risks information loss on critical habitats (see Section 4.5) and on spatial organisation. Spatial organisation of habitats can be very important to fish population dynamics as fish utilise different habitats at different times (e.g. Kocik & Ferreri 1998).

Technology is available to rigorously define habitat variability between river segments and to determine representative river reaches. Hardy & Addley (2001) surveyed over 900 km of river channel at 0.5 m resolution using satellite and airborne remote sensing in a 1-week period, which provided delineations of riparian and fish habitat. A GIS was used to present a visual representation of the study reach in terms of its component spatial distributions of depth, velocity, substrate, cover, distance to features etc. (a GIS can also be used to

Figure 2. Transect locations in part of a representative reach (after Bovee & Milhous 1978).



derive a number of statistical characterisations to demonstrate the efficacy of the modelling approaches).

Contrary to recommended practice, IFIM studies in New Zealand have often neglected to objectively determine study reaches. It follows that there is no theoretical justification for the subjectively chosen representative reaches nor is there a valid basis for extrapolating survey results (R.J. Barker, University of Otago, Department of Mathematics and Statistics, pers. comm.).

4.4 MESOHABITAT TYPING = HABITAT MAPPING

Morhardt et al. (1983) developed habitat mapping (termed ‘mesohabitat typing’ by Bovee (1997) and ‘physical biotopes’ in the European literature) for situations where regularly repeating patterns (such as a pool-riffle-pool-riffle sequence) are not evident (i.e. the distribution of mesohabitats is random or inconsistent). Habitat mapping involves the definition and explicit inventory of the proportions of mesohabitats in a segment.

The procedure is summarised by Bovee et al. (1998).

- Mesohabitat types are defined for the stream under investigation.
- An on-site inventory is conducted to determine the proportion of the segment represented by each mesohabitat type.
- Two or more mesohabitat reaches representing each type are selected at random.

- Transects are established to represent the mesohabitat type.
- Transects in each mesohabitat type are weighted according to the proportion of the mesohabitat type in the segment.
- The segment is represented by all transects from all of the mesohabitat types, combined into a single data set.

A cautionary note is sounded in the American Fisheries Society manual of common methods of aquatic habitat assessment (Bain et al. 1999): ‘... in the past two decades fisheries and natural resource agencies have increasingly employed habitat-based approaches for resource inventory and assessment. Habitat is now the basis of many forms of species management, mitigation planning, environmental regulation, and impact assessment ... However, the validity of habitat-based management rests on accurate definitions and measurements’.

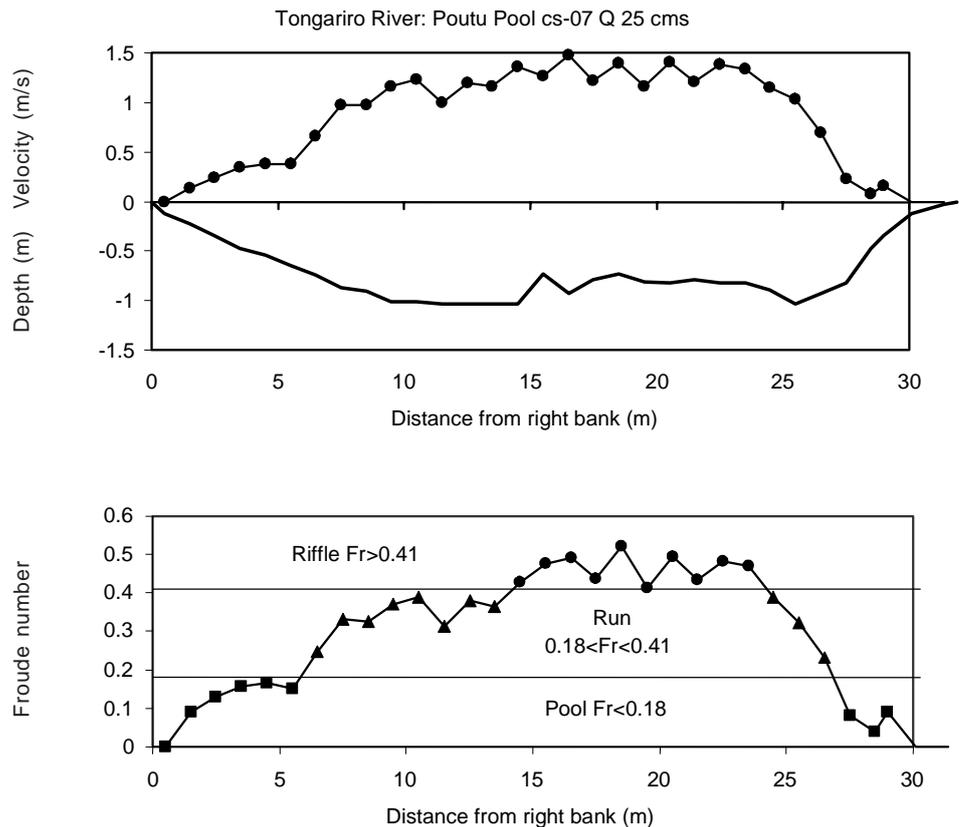
In practice, North American studies usually use detailed classification schemes to delineate habitat units (Arend 1999a, 1999b). Fast water habitats (falls, cascades, chutes, rapids and riffles) and slow water habitats (including a variety of scour pools, such as trench pools along an erosion-resistant bank, convergence pools where two channels meet, lateral scour pools on the outside of a bend, and plunge pools downstream of a rock ledge or tree trunk) are distinguished. Also, various types of dam pool are recognised (e.g. abandoned channels, backwaters and those behind obstructions).

Several independent studies have shown good relationships between habitat structure and associated fish assemblages (e.g. Beschta & Platts 1986; Frissell et al. 1986; Rabeni & Jacobson 1993).

No such classification schemes are widely used in New Zealand, and in much of the New Zealand fisheries literature mesohabitat terms such as riffles, rapids, cascades, glides, runs and pools have been used inconsistently (Hudson 1998, 2001a), but the latter is also a common problem internationally (Arend 1999b). In fact, the typology of New Zealand rivers is simplistic (MFE 1998) and the GIS-based classification of river types developed for New Zealand (Snelder et al. 1999) does not involve field surveys (which are often necessary to discriminate and delineate mesohabitat units). Rather, RHYHABSIM uses a hydraulic criterion to define pools, runs or riffles (Jowett 1993a; Jowett et al. 1996), a well recognised method that can be applied consistently. Unfortunately, these units make up only a small sub-set of habitat types (Arend 1999b), and our calculations show that the Froude number delineations of habitat types have often been inconsistent with standard definitions (e.g. see Fig. 3). Data from another region of New Zealand, Otago, also revealed little correspondence between the habitat units determined in field surveys (ORC 2001) and RHYHABSIM Froude classifications (Hudson 2001a). This is consistent with a study by Buffagni et al. (2000), who found that Froude number was not a useful descriptor of the physical variation between habitat types in the River Ticino, Italy. It seems that retrospective classification of habitat units using the hydraulic criterion in RHYHABSIM may be incorrect in many circumstances.

Further, the use of a hydraulic criterion in habitat mapping can make it difficult to determine the lengths of habitat because pool-riffle-run habitats often do not have well defined transitions (e.g. Jowett 1993a), and habitat units change over time (Giberson & Caissie 1998) and with flow (Heggenes 1996;

Figure 3. Habitat delineations using Froude number are inconsistent with accepted habitat descriptions (modeled data from Jowett et al. 1996).



Hilderbrand et al. 1999). Where the term pool, run or riffle is used to describe a structural unit of a river system, water surface slope or some other measure of longitudinal bed profile may be the most appropriate variable for objective classification (Jowett 1993a).

For any stratification system by habitat unit to be useful, independent observers must be able to classify habitat units objectively and consistently (Roper & Scarnecchia 1995), and be able to estimate dimensions of the habitat units once the units have been identified (Hankin & Reeves 1988). Good training and guidelines are essential (Roper & Scarnecchia 1995; see Section 8).

As for representative reach surveys, New Zealand habitat mapping studies have generally not rigorously rationalised study site selection and so there is no statistical foundation for extrapolating from subjective habitat mapping survey results. They have also risked missing critical habitats and underestimating the importance of the spatial organisation of habitat components. In many studies, insufficient site information has been provided to allow replication of the study.

4.5 CRITICAL REACHES

Critical reaches are areas in the stream that are particularly sensitive to change in flow (e.g. riffles and braid bars) and/or are critical to the success of a particular species' life stage (Bovee & Milhous 1978). A typical example where one reach meets both criteria is the riffles in the braided Ashburton River,

which are sensitive to flow changes and are impediments to salmon migration at low flow (Hudson 2003).

Critical sections are surveyed if habitat bottlenecks can be identified (Stalnaker et al. 1995) (e.g. impediments to fish migration, limitations for spawning or juveniles). Hayes (1998) surveyed only runs and riffles in the Waimea/Wairoa and Wai-iti Rivers, New Zealand, because the extensive pool habitat is relatively independent of flow. The riffles and runs are migration impediments and trout drift-feeding habitat, respectively. In studies of the Rangitata River, particular attention was given to the role of streamflow in the creation and maintenance of seep channels (Hudson 2001c, 2003) because of their high productivity, utilisation by small fish and wading birds (Sagar 1983; Digby 2001) and possible dewatering at low flow.

MESC (2001) noted that the assumed role of a particular mesohabitat type (e.g. pool, riffle, backwater) as the limiting factor to success of the species and/or life stage should be verified. For a single species, different mesohabitat types may be limiting to different life stages at different times of the year, and different mesohabitat types may limit other species. Regardless, it is important that the study site(s) represent(s) the full range of mesohabitat types present in the study area. This typically involves the use of a representative reach or a habitat mapping strategy.

In our experience, IFIM studies in New Zealand have seldom explicitly examined critical reaches or habitats. Hayes (1997) noted that in the Ashburton River it is uncertain whether past surveys included shallow water habitats (riffle crests) that inhibit fish passage. Subsequent surveys found impediments occurred (Hudson 2003). In the Ngaruroro River, a number of habitats, which may have been critical, were avoided (Wood 1997); 'A number of fast deep runs were seen but these were inevitably too fast and deep to be safely gauged ... Areas of highly developed braiding were avoided on the assumption that at low flows the river would reduce to a single channel. It was therefore considered more appropriate to evaluate reaches which were already in single channel form'.

4.6 TRANSECT LOCATION

As outlined in Section 3.3, in 1D modelling, habitats are described by transects across a stream channel and microhabitat variables are measured (water depth, velocity and substrate-cover) at numerous points. Considerable emphasis is placed on transect locations to determine hydraulic controls and to describe habitats (Bovee 1978, 1997). Transects are required at hydraulic controls if a water surface profile program is used to calculate reach hydraulics (dark lines in Fig. 2), but may be omitted if hydraulics are calculated from measured depths and velocities. The reason for the emphasis on transect location is simple: too few and/or incorrect locations and descriptions of transects produce WUA relations that are meaningless (Williams 1996; Bovee 1997).

(Recall that in 2D modelling, an explicit aerial representation of the stream is used rather than isolated transects that yield data for extrapolation; Waddle 1998.)

In stratified random sampling of representative reaches, transects must be relatively closely spaced. Simonson et al. (1994) showed that for wide streams (wider than 35 m), 20 transects spaced every two mean-stream-widths apart are required to achieve mean values within 5% of the true value 95% of the time.

For habitat mapping surveys, Morhardt et al. (1983) pointed out that ‘... it may be desirable to have more than one transect in each habitat type, but that decision should be an intentional part of the sampling design’. In their example of ‘one transect in each habitat type’, Morhardt et al. (1983) delineated components of a pool, and placed transects in the head, middle and tail of the pool. They also placed transects in the chute immediately above the pool, in a gravel run and in high- and low-gradient boulder runs. This is similar to sampling ‘physical biotopes’. Padmore (1998) described transects spaced at 5–10 m intervals, depending on hydraulic variation, to describe one complete biotope sequence. This contrasts with some New Zealand applications, where transects have been placed hundreds of metres to kilometres apart. In fact, habitat simulation applications in New Zealand often have had problematic transect placement.

- Critical habitats (e.g. impediments to fish passage, and mid-channel bar reaches) have often not been explicitly surveyed.
- Transects were often very narrow relative to the active river channel width, and very widely spaced apart.
- Representative reach sections have been short (a few channel widths in length) rather than the 10–15 channel widths common internationally.
- Habitat mapping transects have been located without rationalisation of the sampling strategy.
- Sections that were too deep to wade, too shallow to jet boat or too shallow to gauge, have been avoided.
- Hydraulically complex sites have been avoided.

These tendencies are not unusual. Commenting on North American work in particular, the developers of IFIM (Waddle et al. 1997) stated: ‘In our experience, many [1D] applications ... avoid important habitat areas in complex channels due to large field data collection and analytical resource requirements. Two-dimensional models provide a means to overcome some of these difficulties’.

Uncertainties in transect data can have severe repercussions. In one study on the Feather River, California, there was equal probability that optimal habitat occurred at 71 m³/s as at 14 m³/s (over a 10-km reach) even when uncertainty in the transect data were ignored (Williams 1996). In other simulations, where transects were weighted by the proportion of mesohabitat, the confidence decreased. When the uncertainty in the transect data is taken into account, the analysis shows that estimates of WUA can become practically meaningless. For these reasons, Castleberry et al. (1996) stated that estimates of WUA should not be presented without confidence intervals.

We are not aware of any comprehensive studies of total errors in habitat simulation nor are we aware of any New Zealand studies which have reported confidence limits, even when uncertainties in the transect data have been ignored.

4.7 MICROHABITAT SUITABILITY

Although other physical variables, if they are hydraulic or structural in nature, can be used to define habitat suitability (Bovee et al. 1998), we restrict our discussion to the four most often used variables (see Section 3.3): depth, velocity, substrate and cover (the later two may be referred to as channel index).

4.7.1 Depth and velocity

In wadeable streams, depth can often be measured with a reasonable accuracy by trained observers (Wang et al. 1996). At some locations depth can fluctuate by several centimetres at constant discharge, but this can be detected and measurements standardised (e.g. the low point of fluctuations). Bed levels often vary considerably around individual cobbles or boulders, potentially generating differences between depth measurements taken using wading rods (having a c. 8-cm-wide base plate) and those using survey rods (having a pointed shaft which can be placed between rocks). Placement should be explicitly described and standardised to avoid generating differences in measurement (these may be in the order of the low flow regime water levels being negotiated).

Velocity normally refers to the mean water column velocity rather than the velocity at the expected location of a fish or aquatic animals in the water column (the nose velocity or focal point velocity). In fact, mean water column velocity may not be the best measure of velocity. Vertical velocity profiles often deviate substantially from the commonly assumed logarithmic profile even in simple gravel-bed river channels (citations in Kondolf et al. 2000) such that the highest velocities are sometimes near the bed. For a relatively straight channel with a tranquil flow appearance, Kondolf et al. (2000) found the velocity at 0.6 depths (the hydrological standard; the mean of 0.2 and 0.8 depth) was generally a small overestimate of the vertically averaged velocity and sometimes it was an underestimate by almost 60%. They suggested that in steep channels with large roughness elements flow patterns would be even more complex.

Nose (focal point) velocities can be simulated in PHABSIM-2 (Bovee 1986), and later versions, but results are highly variable (Gan & McMahon 1990; Bovee et al. 1998; Milhous 1999b). In a cobble-bed stream Milhous (1999b) reported that most calculated nose velocities were acceptable (less than 30% error), but that errors in the other 15% of cases exceeded 100%.

Orth (1987) and Scott & Shirvell (1987) explicitly suggested that mean water column velocity is inappropriate for describing fish habitat preference, arguing that conditions at 0.6 depth are not what fish perceive or respond to when selecting positions. By changing the velocity measurement point from the hydrological standard to 15 cm above the streambed, the velocity may be closer to what fish perceive or respond to when selecting positions. However, the ecological significance of a 15-cm depth for velocity measurements is questionable. In one study (Milhous 1999b), twice as much WUA was calculated to be available for adult rainbow trout when nose velocities estimates, rather than mean column velocities, were used.

Bovee et al. (1998) also suggest bed shear stress as an alternative microhabitat variable. This may be appropriate for New Zealand conditions because many New Zealand native fish species are small (less than 150 mm long) and benthic (McDowall 1990). Bed shear stress can be modelled in the latest version of PHABSIM (MESC 2001) or externally using outputs from RHYHABSIM.

4.7.2 Substrate

Substrate is described in terms of organic detritus, various size ranges of sediment particles and rock. Bovee & Cochnaur (1977) started with eight categories for their channel index; adjustments were made by Bovee (1982) and the classification was revised in the 1986 IFIM manual (Bovee 1986). The current IFIM manual (Bovee 1997) uses the Bovee (1986) classification (Table 3).

The increase in the number of size classes was biologically driven (e.g. chinook salmon use a narrow range of substrates for spawning; Platts et al. 1979), and the size class breaks were standardised to conventional geomorphic-engineering definitions.

In IFIM analysis the emphasis has usually been on describing surface materials. Typically, surface materials are significantly coarser than subsurface materials, but veneers of fine material may overlies coarse material. Logically, both the surface and subsurface material should be considered in habitat assessment. The surface material provides the hydraulic roughness of the channel for hydraulic calculations and provides shelter for animals. The subsurface materials are also an important habitat in which many bottom-dwelling fish reside, invertebrate communities proliferate (citations in Collier & Scarsbrook 2000) and both fish and invertebrate egg incubation and hatching occurs (Reiser 1998).

Description of substrate for PHABSIM modelling is problematic. Kondolf (2000) indicated there is no evidence that visual estimates of substrate size classes, as normally undertaken for PHABSIM, are reproducible between investigators and

TABLE 3. SUBSTRATE TYPES AND SIZE CLASSES USED IN IFIM.

SUBSTRATE TYPE & SIZE CLASS (mm)	IFIM MANUALS (BOVEE 1986, 1997)	NEW ZEALAND PRACTICE (JOWETT 1996)
Organic debris	Logs, branches, leaf litter	(1) Vegetation
0.0002 to 0.004	Clay	(2) Silt/Mud (<0.06 mm)
0.004 to 0.06	Silt	
0.06 to 2	Sand	(3) Sand
2 to 4	Very fine gravel	(4) Fine gravel (2 to 8 mm)
4 to 8	Fine gravel	
8 to 16	Medium gravel	(5) Gravel (8-64 mm)
16 to 32	Coarse gravel	
32 to 64	Very coarse gravel	
64 to 128	Small cobble	(6) Cobble (64-256 mm)
128 to 256	Large cobble	
256 to 512	Small boulder	(7) Boulder (>256 mm)
512 to 1024	Medium boulder	
>1024	Large boulder	
Bedrock	Four classes	(8) Bedrock

other studies found poor precision in substrate determination (Platts et al. 1983; Wang et al. 1996). Changes in substrate over time need to be recognised (Giberson & Caissie 1998).

In New Zealand practice, similar substrate characterisation problems are evident. Inconsistent definitions of substrate size classes have been used (e.g. cobbles have been classed as boulders; Hudson 2001a). Substrate has often been assumed, or reported, to be consistent across the channel (Mosley 1983; Glova & Duncan 1985), but this is unlikely for gravel-bed rivers (Powell 1998). Mosley (1983), using visual observation to classify substrate sizes for habitat simulation in four braided gravel-bed rivers in Canterbury, reported: 'No lateral variation in bed sediment character was apparent across the channels surveyed, so that as discharge and channel width increased, the newly inundated stream bed was composed of sediment with a grain size distribution broadly similar to that already under water.' However, in a resurvey of one river, Mosley & Tindale (1983) noted, 'There is great variability of both surface and bulk sediment in the study reach [Spatial variability] ... is broadly consistent with models of grain-size variation'

In New Zealand habitat simulations, about half the categories of sediment size classes are used (Table 3) compared to international practice, resulting in the clustering of size distribution data into broad categories in the HSC. Habitat availability may be overestimated or underestimated as an artefact of the size classes used. According to Hudson (2001b), preference indices ranged from 0.36 to 1.0 for the cobble class for small longfin eels, a range that spans from below average to optimum habitat in modelling. He suggested that a larger number of substrate classes would have reduced the variability within each class to better reflect the observed relation between substrate and use.

Consistent and defensible measures of bed material (substrate) composition are required to avoid many of the problems apparent in the fisheries literature (Kondolf 1998, 2000). The hydrodynamics of bed-sediment interactions and ecological consequences are complex (e.g. Sear 1993). For example, brown trout avoid spawning in loose clean gravel, and spawn in gravels with some interstitial silt (McDowall 1990). Deposition of a layer of fine material over the spawning bed may be beneficial (if it prevents deposition of material in the egg pocket) or detrimental (if it impedes flow through the redd or emergence of fry) (Bjornn & Reiser 1991). Simple indices of bed material do not describe these conditions (Kondolf 1998, 2000). The preferred approach would be to use a continuous function of sediment size in the computation of habitat availability.

In addition to their particle size, the embeddedness of cobble and boulders can be assessed. Embeddedness is an index of the degree to which these larger particles are surrounded or covered by finer sediment (Platts et al. 1983). As embeddedness increases, biotic productivity of the substrate is considered to decrease (e.g. Lapointe & Payne 1996; Milhous 1996). However, embeddedness has not been used in New Zealand habitat descriptions or habitat simulation, although embeddedness is thought to be important (e.g. for eels: D.J. Jellyman, NIWA, pers. comm.; and wrybills and other wading birds: K.F.D. Hughey, Lincoln University, pers. comm.). Embeddedness should be investigated as a habitat suitability index; and to determine flushing flow requirements.

4.7.3 Cover

Refuge elements like macrophytes (weeds), woody debris, undercut banks and river shading (Cullen 1994) are important habitats of fish (Orth & Maughan 1982). For example, comparisons of the density of adult brown trout to percentage habitat area available in two creeks in California showed that over 90% of the fish were found in less than 2% of the total habitat available (Kershner & Snider 1992), specifically in lateral scour pools with complex woody debris. Cover is also important for New Zealand's native species (McDowall 1990; Hanchett 1990; Rowe et al. 1992, 1999; see Section 6.2.3), but has not been included in New Zealand IFIM studies.

While cover is recognised as important, there is little agreement on what constitutes cover and a poor understanding of how it can be quantified for use in IFIM studies (Scott & Shirvell 1987). Furthermore, cover changes with streamflow (Orth & Maughan 1982). Bovee (1986) recommends using conditional criteria. For example, some fish use shallow water if there is overhead cover, but they will only use stream cells in deeper areas where cover (or surface turbulence) is absent. This can be depicted in PHABSIM using two depth-suitability curves, with and without cover.

In 2D modelling, large cover elements can be included in habitat simulations (Leclerc et al. 1995; Bovee 1996; Ghanem et al. 1996; Crowder & Diplas 2000; Waddle et al. 2000), but these may be too large-scaled for many species.

4.8 HYDRAULIC-HABITAT SIMULATION

There are numerous criticisms of statistical and hydraulic methods used during the hydraulic modelling process (summarised by Mathur et al. 1985; Scott & Shirvell 1987; Kondolf et al. 2000). PHABSIM and RHYHABSIM are appropriate for only steady, gradually varied, subcritical flows. These conditions may not be met in steep streams, during hydro-peaking flows or at low flows when bedrock is exposed (McMahon 1992). As models are difficult to calibrate (Osborne et al. 1988; Ghanem et al. 1996), in many applications complex hydraulic sites, which may be biologically significant, are avoided (see Sections 4.5 and 4.6).

Gan & McMahon (1990) stated that the potential for 'fudging' results within PHABSIM is considerable. The WUA-flow relationship obtained depended on a broad range of choices exercised within PHABSIM for a given set of inputs. Not only can different flow simulation models be chosen, the models themselves have a range of simulation options. RHYHABSIM (Jowett 1996) (used in New Zealand to simulate river hydraulics and habitat as WUA) has fewer options for modelling hydraulics (Gordon et al. 1992). Although modelling results are easier to replicate it is uncertain how a broad range of hydraulic conditions can be adequately modelled. Only once the hydraulics are adequately calibrated, should the simulation proceed (Bovee 1997).

To describe flow patterns in river channels, 1D models break the reach into discrete cells, each having a uniform depth and velocity (Bovee 1978). While the length of the cells may be tens or even hundreds of meters long (i.e. from transect to transect), the width of each cell may be relatively narrow (typically transect point measurements are 1–2 m apart for streams and small rivers).

Frequent measurements across a channel are a hydrological convention. The implication of this survey strategy is that cross channel variation is of greater importance than variations along the channel.

In reality flow patterns are complex and vary both spatially and temporally across and along the channel. Wakes and high velocity gradients occur around boulders, root wads, debris and bed formations. Boulders and rock clusters create low shear stress zones that play an important role in determining the diversity of periphyton (slime and filamentous green algae) and invertebrates in a stream ('microcluster refugia'; Biggs et al. 1997). These local variations in flow, such as velocity gradients and transverse flow, within a cell cannot be simulated with 1D models (Crowder & Diplas 2000). This was evident in Poutu Pool, Tongariro River, where the 1D model predicted smooth streamlines between the seven cross sections (Jowett et al. 1996). However, flow patterns in this reach are highly disrupted with boulders creating lees, eddies and chutes (unpubl. data).

Even within relatively uniform channels modelling may not describe the flow variability particularly well. Kondolf et al. (2000) described the findings of Bartz (1990). He compared the measured and modelled velocities for each vertical calculated by PHABSIM and found mean errors ranging from 4.6% to 12.8% and standard deviations from 29.6% to 42.7% for small and large streams.

As mentioned in Section 3.3, alternative approaches involving 2D and 3D models and hydrological equations may lead to greater accuracy and resolution in hydraulic simulation, and allow modelling flows around bed clusters (Biggs et al. 1997) and at the bedform scale, which are important to fish (Davis & Barmuta 1989; Shirvell 1989). 2D models have not been used routinely in New Zealand but their use has increased in North America (Leclerc et al. 1995; Crowder & Diplas 2000). The flexible mesh of River2D allows more detailed flow simulation in complex areas (e.g. around large boulders, riffles, deep channels along near vertical banks) compared to the fixed grid modelling that has been used in New Zealand (see Section 3.3; also, Waimakariri River: Duncan 2001).

Bovee (1996) was of the opinion that the advantages of 2D hydraulic modelling became apparent in spatially explicit habitat models. In detailed surveys and 2D modelling, fish abundance was significantly correlated with indices of habitat heterogeneity, suggesting that for some species habitat structure may be as, or more, important than habitat availability (Stewart 2000). Kocik & Ferreri (1998) suggested that functionally discrete habitat units occur within rivers and that the spatial organisation of these units influences fish population dynamics.

A major issue in modelling is that fish appear to respond to features in their hydraulic environment, such as velocity gradients, over very small longitudinal scales (Kondolf et al. 2000). For example, salmonids may hold in the flow separation zone downstream of a boulder, with minimal tail beat, while the current only millimetres overhead is as high as 60-70 cm/s (Bachman 1984). Hayes & Jowett (1994) observed that brown trout dart from velocity shelters into fast water to feed. These vertical velocity gradients can only be crudely approximated (Kondolf et al. 2000) and small-scale features, such as boulders, root-wads and other obstructions, can be simulated in the 2D flexible mesh

models, but the survey requirements are onerous (Crowder & Diplas 2000; Waddle et al. 2000).

Generally, there is a major disparity between the scale at which habitat is modelled and the scale at which HSC have been derived. Bult et al. (1998) suggested that current habitat modelling may be improved by more explicit use of time and space scales. Hayes et al. (in press) illustrated the conditional nature of habitat selection on multiple scales which may have profound impacts on the nature of HSC. Research is required to establish appropriate scales for measuring and modelling, which may vary between species and life stages. As a starting point, spatial resolutions of 1–2 m may be appropriate for salmonids (e.g. Grant & Kramer 1990; Hill & Grossman 1993; Hughes 1998).

5. Habitat suitability criteria (HSC)

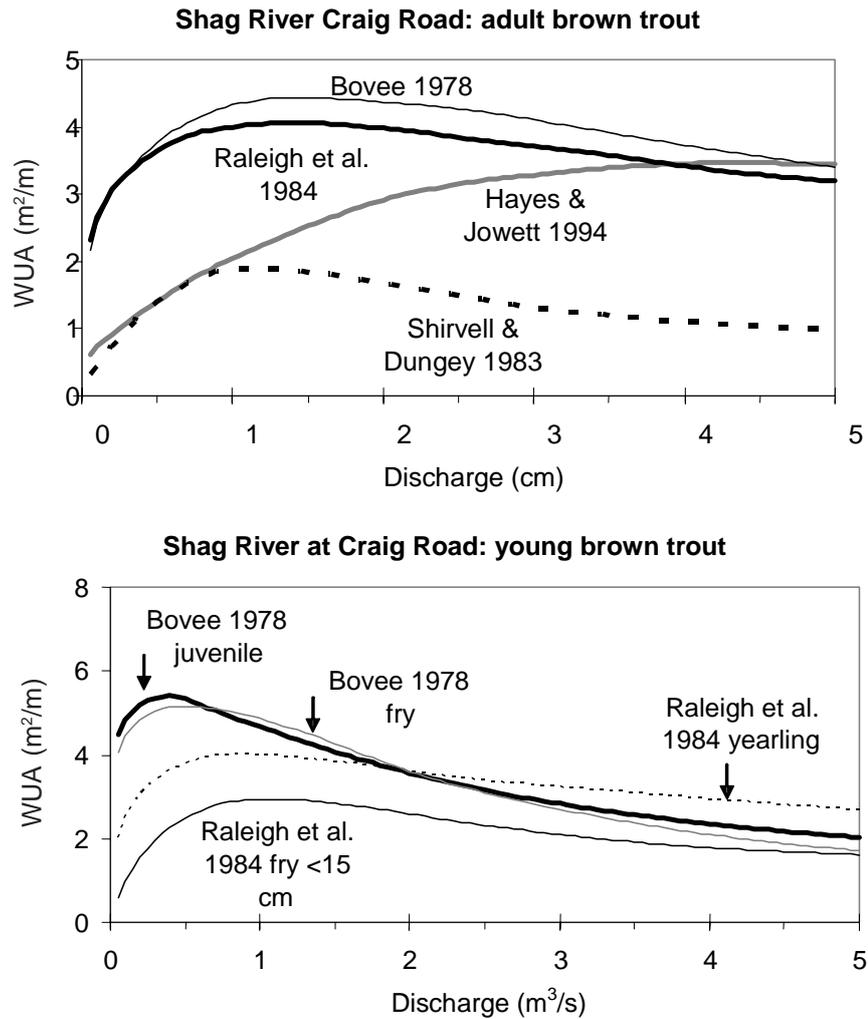
5.1 APPLICABILITY AND TRANSFERABILITY

‘The greatest single constraint to the proper implementation of IFIM analysis is the use of accurately derived habitat suitability curves ...’ (Gore & Nestler 1988). Use of curves that do not accurately characterise the utilised or preferred habitat for a species can cause significant error (Bozek & Rahel 1992; Thomas & Bovee 1993; Glozier et al. 1997).

In IFIM analysis, habitat suitability curves are obtained from a library of species curves (e.g. Bovee 1978; Jowett 1996), developed by an expert panel (Bovee et al. 1998) or, preferably, are developed in the river in question (Bovee 1986; Gore 1987). The transferability of the curves should be tested in the study reach (Bovee et al. 1998) and this has been a policy in some jurisdictions (Stalnaker 1994). If transferability tests are not undertaken it is uncertain which, if any, habitat suitability curves are appropriate since there are often considerable differences in the shape, position and amount of habitat predicted depending on the HSC used and life stage considered (Fig. 4).

Few studies have been published involving transferability tests or HSC verification (e.g. Armour & Taylor 1991; Thomas & Bovee 1993; Stalnaker 1994). The reliability and robustness of the test procedures have been extensively evaluated in several rivers. Glozier et al. (1997) tested the hypothesis that habitat suitability curves for a widely distributed North American forage minnow were similar in two rivers in Canada. The work involved evaluation of library and local curves and application of Thomas & Bovee’s (1993) transferability tests for habitat suitability curves. The authors found these tests did not provide a definitive answer on HSC transferability—the tests often failed because the test statistic was not applicable to the raw data.

Figure 4. Shape and position of the WUA-flow relationship with the HSC and by life stage (modeled from data in ORC 2001).



5.2 SAMPLING PROBLEMS

Preference curves are meant to be obtained from unexploited streams at carrying capacity (Bovee 1982; Mathur et al. 1985). However, such conditions are rare at best, especially for high-quality habitats, and are difficult to predict prior to investigations (Mathur et al. 1985) and difficult to assess. If reaches that are sampled to determine the curves are not at carrying capacity, the resulting curve may be flawed and one should expect variation in curves determined at different times and locations. Mathur et al. (1985) showed that curves developed for adult smallmouth bass have completely different shapes and the optimum 'preferred' or 'suitable' depth and velocity differed by an order of magnitude between curves for each variable. Differences in curve shapes have also been reported by Bozek & Rahel (1992) and Glozier et al. (1997).

It would seem to be physically difficult to obtain unbiased estimates of fish habitat preferences, because fish are highly mobile and respond rapidly to any kind of intrusion. This is even more problematic where turbidity limits observation (Gore & Nestler 1988). Several sampling protocols have been developed to reduce bias but, according to Bovee et al. (1998), there is no unbiased way of collecting habitat use data.

Further, preconceived notions, and the method of sampling, may lead to erroneous results, as suggested by a study by Groves & Chandler (1999). Using remote underwater video to describe spawning habitat use by chinook salmon in the Snake River, Colorado, they found spawning depths ranged from 0.2 m to 6.5 m, substrate level water velocity to be 0.1–2.0 m/s (mean column velocity of 0.4–2.1 m/s) and substrate size to be 25–150 mm diameter. These findings greatly expanded current criteria used to model spawning habitat availability for these fish in larger rivers.

Bias can be even more difficult to avoid for our primarily small (less than 150 mm), cryptic, benthic and often nocturnal New Zealand native species (McDowall 1990). For example, while Jowett & Richardson (1995) provided valuable information on native fish habitat use, only limited ranges of habitat types (runs and riffles) and conditions (depths of 0.08–0.8 m, 80% being less than 0.4 m; substrate up to 192 mm) were sampled during the day, in summer. Their conclusion, that the more common native species have a well-defined preference for relatively shallow water habitats (Jowett & Richardson 1995; Richardson & Jowett 1998), has not been supported by other observations (e.g. Main et al. 1985; Taylor & Main 1987; Taylor 1988; Hayes 1995; Chadderton & Allibone 2000). Rather, native species have been found to occupy a range of habitats including deep water (Taylor 1988; Chadderton & Allibone 2000) and fast flowing yet relatively shallow waters, which cannot be sampled by conventional methods such as electro-fishing and diver observation (Hayes 1995). Chadderton & Allibone (2000) provided strong evidence for augmentation of standard sampling procedures (electro-fishing) with traps and fyke nets in deeper waters.

Timing of sampling may also be problematic at various scales. Instantaneous measures of physical habitat characteristics are not necessarily related to instantaneous measures of fish population size, because biological responses (in terms of population dynamics) are slower than other flow-related phenomena (Williams et al. 1997). Hence, population size at any particular time is determined by limiting factors that may have been operating in the past but no longer are (for example, scouring or dewatering of redds, stranding of the young of the year, and lethal and sub-lethal temperatures).

Specifically, samples taken during summer low flows may not adequately describe HSC since the hydraulic characteristics of a habitat changes with flow, and habitat selection may change with season (e.g. Orth & Maughan 1982). Temperature-mediated competitive or predatory interactions may cause shifts in habitat use. Further, water clarity may change with flow, such that increased turbidity causes a switch in diet from drift to benthic prey, and/or a change in preferred microhabitat to slower current speeds to compensate for reduced prey detection (Metcalf et al. 1997). In addition, habitat preferences of some species are flood-dependent (e.g. banded kokopu and short jawed kokopu both require elevated flows for egg deposition and hatching (Charteris et al. 2003). Species such as lamprey may migrate when the water is dirty as a result of flooding (McDowall 1990).

On a smaller time scale, transect sampling on a river or stream during the day may not necessarily give a representative indication of where fish are located in relation to physical habitat variables. Diel habitat shifts are well documented for

salmonid species (e.g. rainbow trout: Hill & Grossman 1993; brown trout: Heggenes et al. 1993; and Atlantic salmon: Metcalfe et al. 1997); wild fish occupied slower currents at night than during the day. Metcalf et al (1997) suggested that these shifts may differ between nights depending on the effects of available light and water temperature on optimal foraging ranges, fish acceleration and sustained swimming speed.

The previously cited study by Jowett and Richardson (1995) typifies the problems in sampling native New Zealand species. Many are nocturnal or partly nocturnal (McDowall 1990) and habitats not occupied during the day may be important feeding habitats at night, which daytime sampling will miss. Diel, ontogenetic and seasonal shifts in activity, and pool and riffle habitat use have been documented in giant kokopu for example (David & Close 2003; Whitehead et al. 2002).

The redistribution of fish seasonally, diurnally and with changing flows, and the fact that the availability of usable area may not have an immediate regulatory effect on the population, are significant considerations for New Zealand surveys. The tendency in New Zealand has been to use data from one-off surveys, which are dominated by single, site-specific samples taken in isolation of past events. The wider relationship between physical and biological variables must be considered, given that these variables can vary with geology, evolutionary history, genetic history, physiological pathway development, ontology, life history, or any combination of these irrespective of available physical habitat (Castleberry et al. 1996; Williams et al. 1997). Recognition of the changes in habitat uses in time and space prompted Bovee et al. (1998) to state 'We cannot simply define microhabitat requirements for the species in general, we must specify the life stages, sizes, activities, and time periods for which our definitions hold'.

5.3 QUANTIFYING HABITAT SUITABILITY

5.3.1 HSC development

The original application of IFIM treated habitat suitability curves as probability functions. The peak of the curve was given a weighting value of one and represented the optimum value of a variable for use by a given life stage of a given fish species. The tails of the curve received a weighting of zero (Bovee 1978, 1982). Mathur et al. (1985) pointed out three problems with this: (1) habitat suitability curves are not probability functions but only ratios based on counts of fish relative to a maximum that was encountered on dates of observations; (2) a curve is only correct when the variables are statistically independent; and (3) two of the key variables, depth and velocity, are known to be related.

This is fundamentally important (Beecher et al. 1997): 'Do depth and velocity preferences determined at one flow predict fish distributions at another flow? If not, then the instream flow incremental methodology (IFIM) will not be useful for evaluating the effect of different flow on fish.' These authors (1995; 1997) claimed to show that depth and velocity preferences are independent of flow for juvenile steelhead. Jager & Pert (1997) and Williams (1997) disputed the

methodology and findings of Beecher et al., and other studies have shown changes in preference for microhabitat variables with changes in streamflow and other factors (see Section 4.7), casting doubt on the ability of simple indices to describe habitat.

Bovee et al. (1998) pointed out that multivariate criteria overcome the problems of assumed independence of variables, but these HSC are not widely used. Further, they produce a single maximum value that does not reflect thresholds. Alternative approaches to develop habitat suitability curves have been proposed. Guay et al. (2000) used logistic regression to construct a multivariate statistical model which produced an index representing the probability of observing fish under specific habitat conditions. The multivariate index had a far greater ability to represent fish distributions and habitat preferences than did habitat suitability curves.

Another promising approach is the application of individual-based models (IBM; first developed by Cheslak & Jacobson 1990). Braaten et al. (1997) proposed using a feeding model as an alternative to habitat suitability curves for drift-feeders. Hayes et al. (in press) have coupled a feeding model with the River2D implementation of PHABSIM.

5.3.2 WUA computation

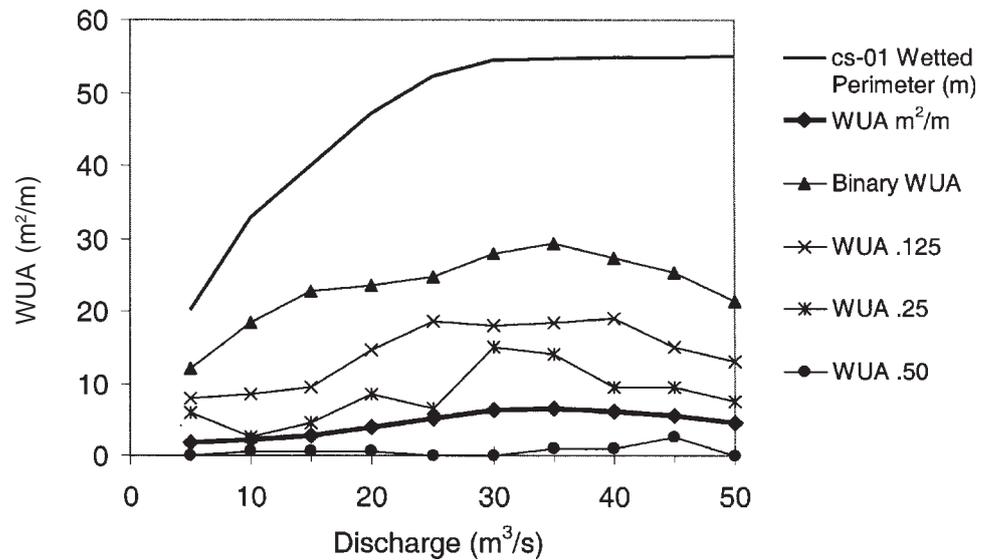
As presented earlier (Section 3.3), habitat suitability curves are combined with the computed cell water depth, velocity and substrate to calculate the WUA, the results varying with the method of computation used. Gan & McMahon (1990) evaluated the results from three methods—the multiplicative (most commonly used, e.g. RHYHABSIM), geometric mean and minimum value approaches—and found that WUA for brown trout fry varied from 145 to 2990 m² per 1000 m length of river, juvenile WUA varied from 663 to 2442 m² per 1000 m and adult WUA varied from 565 to 2167 m² per 1000 m for the upper Tongariro River.

In addition, binary criteria (Bovee et al. 1994; Bovee et al. 1998) have also been used for these calculations. For a given discharge and preference, WUA values can vary by an order of magnitude (Milhous et al. 1989; Gan & McMahon 1990; Kormann et al. 1994; Bovee et al. 1998; Milhous 1999a). In addition, both the shape and position of the WUA-flow relationship can change with the method of computation (Fig. 5) so it is crucial that confidence intervals be presented for all WUA-discharge relationships before PHABSIM or RHYHABSIM results are presented to decision makers (Castleberry et al. 1996; Williams 1996).

5.3.3 Biological meaning of WUA

WUA implicitly considers each habitat unit as biologically equivalent (Bovee 1982) but, as pointed out by Orth (1987) and Scott & Shirvell (1987), large areas of less than optimum habitat do not have the same productive capacity as small areas of optimum habitat. Thus, as in Figure 5, the binary score may be an optimistic prediction of suitable habitat (over half the channel width). The actual optimum habitat may occur at a discharge of 45 m³/s, with an extremely small area of high quality habitat (a combined score of 0.5, with a WUA of 2.5 m²/m of channel, i.e. about 5% of the width). Further, several combinations of

Figure 5. Comparison of WUA calculation using multiplicative (RHYHABSIM), binary and threshold methods for spawning rainbow trout in the Judges Pool, Tongariro River (modeled from data in Jowett et al. 1996).



depth, velocity and substrate can give the same amount of WUA, none of which may support a similar biomass (Mathur et al. 1985).

The big issue in interpretation is which approach (if any) is correct? As stated by Gan & McMahon (1990): 'Of greater concern [than hydraulic modelling options] are the options producing variable results in habitat computations. There are no techniques that validate the results and hence their acceptance must rest entirely on the realism of the biological assumptions that are made.' Shirvell (1989) found that c.70% of the spawning area actually used by chinook salmon in the Nechako River, British Columbia, was predicted to be unusable, while 87% of the area predicted as usable has never had recorded use. Similarly, Geist & Dauble (1998) reported aggregation of chinook spawning redds in definite clusters even though suitable spawning areas were widely distributed. These clusters tended to occur in areas with complex channel patterns rather than where the channel was straight and simple. Further, the patchy distribution in relation to available depth, substrate and velocity suggested that the fish have relatively specific spawning habitat requirements that were only partially explained by microhabitat characteristics used in habitat simulation.

Similar discrepancies have been found on a bedform scale (areas of square metres to tens of square metres). From the above study on the Nechako River, Shirvell (1989) noted spawning exclusively on the downstream side of bedforms and spawning in extremely limited areas with groundwater upwelling, although apparently suitable habitat was available elsewhere. Trials showed that PHABSIM predicted 210% to 660% more spawning habitat was available than historically had ever been used.

Habitat simulation assumes that areas of the stream not occupied by fish are useless, but these areas may still be critical to the fisheries. For example, Jowett (1992) found food production was critical for brown trout abundance in New Zealand. These areas may be dismissed as unproductive if simulation models focusing on one or two species are used as the only means of assessing available instream habitat. Also, the focus on low-flow habitat use can be problematic.

For example, New Zealand backwater areas may be important flood refugia but may only be used at times of high flow. We again emphasise the importance of testing these assumptions and taking a broader view before relying on IFIM to specify a minimum acceptable streamflow. A mesoHABSIM analysis (Parasiewicz 2001) would be instructive in showing the dynamics of critical features such as seepage channels (e.g. Hudson 2001b, 2003).

A fundamental assumption in physical habitat simulation in IFIM is that there is a positive relationship between potential fish biomass and WUA. The MFE (1998) guidelines state (without citations) that studies have found correlations between habitat availability and animal abundance for many species. A frequently cited example is the 'brown trout model' of Jowett (1992).³ A positive relationship has also been found in some streams in the United States (e.g. Conder & Annear 1987; Milhous 1999a), but there have been many more observations of poor or negative correlations (Orth & Maughan 1982; Mathur et al. 1985; Irvine et al. 1987; Orth 1987; Pert & Erman 1994; Shirvell 1994; Zorn & Seelbach 1995; Bourgeois et al. 1996). Indeed, Scott & Shirvell (1987) suggested that the frequency of a positive linear relationship between WUA and fish abundance or biomass is so low that it may be due to chance alone. Orth (1987) stated that a consistent relation between WUA and fish biomass cannot occur because instantaneous counts of fish are related to past habitat limitations for any life stage (see Section 5.2). Hence, instantaneous estimates of suitability should not be related to instantaneous population size (Kormann et al. 1994).

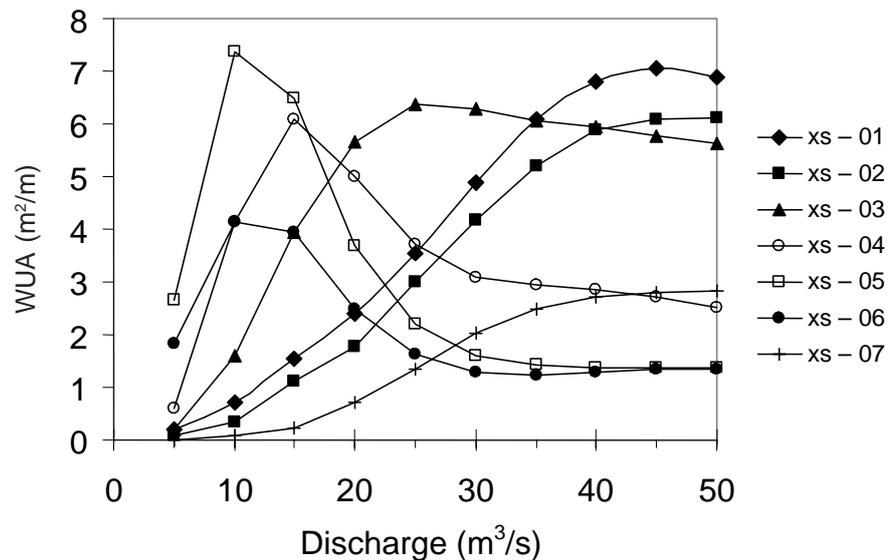
Further, there is a broad range of other important factors determining fish abundance (see Section 4 and footnote 3). According to Pert & Erman (1994) describing fish populations in terms of biomass may yield a false interpretation of optimal habitat for the population as a whole. Jowett (1992) recommended that the relationship between WUA and fish abundance should be demonstrated before the method can be considered valid for assessment of instream flow requirements.

A lack of a relationship, or a weak relationship, between WUA and fish abundance is not surprising. In fact, the developers of IFIM physical habitat simulation (Milhous 1999a) recently stated: 'Analysis of the relation between physical habitat and the populations of aquatic animals shows that the physical habitat is a necessary but not sufficient condition for a viable population of aquatic animals.... there are many interactions between species, life stages, or other variables that influence the state of the ecosystem that are not modelled....'

The aggregation of the suitabilities of all the cells into a single index of habitat availability (WUA m²/m) contains no location information. It also masks the considerable variation that often occurs between individual transects in a reach (Fig. 6). One advantage of 2D modelling is that the individual suitability indices, and combined (weighted) indices, can be compared with observed fish locations on the aerial representation of the stream.

³ Nineteen variables were positively correlated and three negatively correlated with trout abundance. In order, highest correlations were with a temperature preference factor, total invertebrate biomass, % sand, winter temperature, % cobble, total caddis fly biomass, then % WUA for brown trout adults. Twenty-eight percent of total invertebrate biomass was explained by % WUA (Waters 1976 food production curve).

Figure 6. Differences in WUA-flow relationship between transects (xs) of the Poutu Pool, Tongariro River (modeled from data in Jowett et al. 1996).



6. Additional considerations in physical habitat simulations

From a biological perspective, a major criticism of physical habitat simulation has been that it focuses on the physical microhabitat variables discussed in Section 4.7 (Scott & Shirvell 1987; Shirvell 1989; Gordon et al. 1992) as the primary determinants of species use. Implicit in the use of the method is the assumption that physical habitat characteristics are limiting to aquatic species (see Section 3.1), although this assumption is not usually stated (Hicks & Reeves 1994) or tested.

Stream communities are structured by a combination of biotic and abiotic factors (Winterbourn 1995). These include the following essential determinants of communities (from Karr et al. 1986), which vary in time and space.

- Water quality: temperature, turbidity, dissolved oxygen, organic and inorganic chemicals, heavy metals, toxic substances.
- Flow regime: water volume, temporal distribution of flows.
- Habitat structure: substrate type, water depth and velocity, spatial and temporal complexity of physical habitat.
- Energy source: type, amount, and particle size of organic material entering stream, seasonal pattern of energy availability.
- Biotic interactions: competition, predation, disease, parasitism.

Several instream biological processes may limit community structure, such as predation, competition, food availability, dominance hierarchies among individuals, territorial boundaries, fish swimming ability, body morphology, disease and density-dependent mortality. All may complicate habitat-population relationships (Castleberry et al. 1996; Hayes et al. 1996; Williams et al. 1997) and may vary in space and time.

The 1D models routinely used in IFIM (i.e. PHABSIM and RHYHABSIM) struggle to account for: habitat structure (Hill et al. 1991); habitat and flow complexity in space and time; channel evolution over time; energy source and influence of biotic interactions (Bovee 1996; Bovee et al. 1998). For the purposes of our review, we have classified criticisms of current IFIM methodology into three broad categories: biotic, abiotic and methodological, with a focus on 1D hydraulic models as have been routinely used in New Zealand. These categories are interrelated, addressing wider ecological issues that should be considered when setting minimum flow requirements.

6.1 BIOTIC FACTORS

Instream biological processes are likely to have important implications for fish migrations, daily movements within a stream and selection of foraging areas by fish. Their effects on habitat utilisation by aquatic organisms should not be underestimated (NRC 1996). Unfortunately, according to Milhous (1999a): ‘A significant limitation is that there are many interactions between species, life stages, and other variables that influence the state of the ecosystem that are not modelled by PHABSIM.’

6.1.1 Competition and predation

Fish locations within a stream may vary among species, and among individuals within a population. Species distribution patterns are often the result of biotic interactions mediated by abiotic conditions (e.g. the effect of temperature on three fish species by Taniguchi et al. 1998). The presence of competing or predatory individuals can influence habitat selection by fish (Orth 1987). However, the role of competition or predation may change spatially and temporally, with temperature, water quality, flow or life stage. Thus, a species may be eliminated by a predator in some areas, but persist in refugia where abiotic conditions exclude or inhibit the predators (e.g. Rahel 1984; Taniguchi et al. 1998).

In New Zealand there is evidence that introduced fishes have caused significant changes in the distribution pattern of the native fish fauna (McDowall 1990; Minns 1990; Townsend & Crowl 1991; Crowl et al. 1992; McIntosh et al. 1994). Many native fishes show little distributional overlap with introduced fishes and are characterised by fragmented populations, which are vulnerable to extinction as a result of chance events (Crowl et al. 1992; McIntosh et al. 1994).

Interspecific competition or predation has also been documented between native species (Main 1988; Chadderton 1990; Hanchet 1990; McDowall & Allibone 1994; Chadderton & Allibone 2000). Thus, flows that force vulnerable species from refugia, or which enable predators or competitors to gain access to such areas, have implications for the survival of some species.

Competition or predation effects may be further complicated by the influence of temperature and/or season. Taniguchi et al. (1998) demonstrated changes in competitive ability with temperature change, with cooler temperatures favouring brown trout, whereas at warmer temperatures chub became competitively dominant. Although we are not aware of any temperature-

mediated responses studies in New Zealand, such results have implications for the timing of sampling and may be important in Central North and South Island rivers where large temperature fluctuations occur. Competitive or predatory interactions between individuals or species therefore may complicate instream flow assessments. Species-specific analyses of microhabitat availability tend to ignore the effects of these interactions under changing environmental conditions such as crowding at low flows.

6.1.2 Fish migrations including diadromy

Diadromy as a specialised form of migration is a dominant behavioural characteristic of New Zealand fish fauna (McDowall 1988). Almost half of the native fauna and nearly two-thirds of the total fauna are diadromous, these species customarily spending part of their life cycles in the sea (McDowall 1993). Migrations within streams associated with spawning have been documented for common river galaxiids (Cadwallader 1976) and roundhead galaxiids (Moore et al. 1999), and are postulated for giant and banded kokopu (McDowall 1990).

Recognition of the existence of obligatory migrations of organisms within stream systems is a critical issue in developing models to explain or predict fish and invertebrate abundance or distributions in New Zealand (McDowall 1993). The 'assumption that fish abundance will reflect habitat suitability (or vice versa) is likely to break down where migration is a critical life history component' if factors affecting migration have not been quantified and modelled (McDowall 1993). McDowall (1993) stated that there are no explicit data on controls of inland penetration by native New Zealand freshwater fish species, but suggested several factors such as distance (e.g. for inanga, common bully); flow impediments (e.g. rapids: for smelt; vertical falls or long cascades: for bluegilled bully, redfinned bully and torrentfish); and lack of suitable habitat upstream (e.g. for shortfin eels).

It is critical that in developing flow requirements, upstream and downstream linkages are maintained. Only where migratory access does not limit population size can these pathways be ignored. IFIM needs to and can take into consideration critical reaches for fish passage, but this appears to have been rarely considered in New Zealand instream flow applications.

6.2 ABIOTIC FACTORS

Macrohabitat is the set of abiotic conditions such as hydrology, channel morphology, thermal regime, chemical properties or other characteristics in a segment of river that define suitability for use by organisms (Bovee et al. 1998). Macrohabitat controls the longitudinal distribution of aquatic organisms (Shelford 1911: longitudinal succession; and Vannote et al. 1980: the River Continuum Concept). Many physical processes respond to changes in flow regimes and have implications for biological communities.

6.2.1 Water quality

It has often been assumed that water quality is not limiting in New Zealand rivers (Biggs et al. 1990; Close & Davies-Colley 1990). However, point and diffuse sources of contamination occur in New Zealand from agriculture, forestry, industry and urban areas (Hoare & Rowe 1992; MFE 1997). Further, flow manipulation might alter chemical water quality (e.g. type and amount of contaminants released from reservoirs) and physical attributes (e.g. temperature, dissolved oxygen) (NRC 1992) as well as assimilative capacity. If these factors are thought to change and potentially affect habitat suitability for the target species, associated models—simulating the effects of temperature and water quality—can be run with PHABSIM (see Section 3.1). Sediment quality has also been modelled for instream flow assessment (ASACE 2002).

6.2.2 Temperature

Modeling flow manipulation effects on stream temperature has been routine in IFIM applications in the United States (Stalnaker et al. 1995). Although water abstractions and low flows can significantly increase stream temperatures in New Zealand rivers (e.g. Ngaruroro River: Grant 1977; Hurunui River: Hockey et al. 1982), the biological effects have not been well documented. Generally known effects of temperature include (ANZECC 1999) influences on:

- the physiology of the biota (e.g. on growth and metabolism, reproduction timing and success, mobility and migration patterns and production)
- ecosystem functioning (such as through changes in the rate of microbial processes and altered oxygen solubility).

In the Australian National Water Quality Management Strategy draft guidelines (ANZECC 1999), reference was made to earlier thermal effluent guidelines that recommended that the maximum permissible change in the temperature of any inland or marine waters should be 2°C. Alternatively, a lesser change (per unit time) could be set, based on effects of long-term exposure.

The preferred temperature ranges and tolerances are known for many New Zealand aquatic species. For example, banded kokopu prefer temperatures of c. 15–18°C, and temperatures around 30°C are lethal (Richardson et al. 1994). Brown trout spawn in winter and egg mortality increases when water temperature exceeds c. 10°C (Scott & Poynter 1991). The optimum temperature range for adult trout is 12–19°C, and the lethal temperature is 25–30°C, depending on acclimatisation temperature (citations in Elliot 1994). Snails, riffle beetles and a few species of caddisfly are particularly resistant to high water temperatures (Jowett 1997), whereas stoneflies are particularly sensitive and are usually restricted to rivers with summer water temperatures that do not exceed 19°C (Quinn & Hickey 1990; Quinn et al. 1994). Temperatures of 24–26°C are lethal to many stream invertebrates (Jowett 1997).

Effects of flow manipulation on stream temperatures should be routinely considered in New Zealand IFIM analysis.

6.2.3 Refugia/cover

The importance of cover for salmonids was introduced in Section 4.7.3. A number of galaxiid species are associated predominantly with native forest in New Zealand (Hanchet 1990; McDowall 1990; McDowall et al. 1996) and may require coarse woody debris for cover and instream habitat formation (Hicks & Reeves 1994; Rowe & Smith 2003). Foods of terrestrial origin are important for several New Zealand fishes (McDowall 1990). For example, the four largest galaxiids—short jawed kokopu, banded kokopu, koaro and giant kokopu—take terrestrial food items and favour waters with strong riparian vegetation (Main 1988; Main & Lyon 1988; Chadderton 1990; Hanchet 1990; McDowall 1990; McDowall et al. 1996; Bonnett 2000; Bonnett & Sykes 2002). Riparian and instream debris may provide food, overhead cover, temperature mediation, flood refugia and shelter (Main 1988; Collier 1995; Chadderton & Allibone 2000).

6.2.4 Groundwater linkages

The hyporheic biotype is a distinct community with a large degree of interaction with surface flows. It can have a rich fauna which as yet has received limited attention within New Zealand (but see Collier & Scarsbrook 2000). Post-graduate research suggested that the zone may also act as refugia for surface-dwelling invertebrates (Scarsbrook 1995; Olsen 1998). However, reduced stream flows may dewater the upper hyporheic, and linkages can be lost through fines deposition that results from lowered stream velocities (Ward & Stanford 1995).

Groundwater outflow areas in riverbeds can also be preferred spawning habitats (Junk et al. 1989). For example, groundwater upwelling is a prerequisite for spawning success of brook trout (Blanchfield & Ridgway 1997) and the roundhead galaxiid is known to spawn near the head of upwelling subsurface flows (Allibone 1997; Allibone & Townsend 1997; Moore et al. 1999). Hence, groundwater-channel interchange may provide important spawning habitat, but may be damaged by dewatering and the build-up of sand and fines on the streambed (Petts & Maddock 1994). It follows that a model not addressing subsurface flow will be seriously incomplete in its evaluation of habitat for spawning (Kondolf et al. 2000). Hence when establishing environmental flows, maintenance of groundwater exchange processes need to be considered, and catered for; again these have been rarely considered in New Zealand applications of IFIM.

6.2.5 Instream flow variability

Early attempts at regulating stream flows usually established a minimum flow, but it was soon recognised that this failed to provide for environmental needs or multiple uses (Stalnaker 1979). As discussed in Section 2, the necessary periodic high flows that move bed material, flush sediment, rejuvenate the floodplain and generally maintain the structural characteristics of a stream channel were most often overlooked. The need for flow variability is evident (as summarised in Poff & Ward 1989; NRC 1992; Poff et al. 1997), yet MFE (1998) has stated: 'To date [New Zealand] water managers have rarely considered flow variability when setting flow regimes. The normal practice has been to set minimum flows and without considering of the duration of low flow'.

New Zealand streams have been considered physically dominated systems in which biological interactions take a secondary role to climatic and physico-chemical factors (Winterbourn et al. 1981; Cowie 1985; Winterbourn 1995). Disturbance is a primary organising factor in many New Zealand streams, which represent physically harsh environments (Winterbourn 1997). More specifically, the frequency, intensity and magnitude of spate and flood events can determine taxonomic composition, biomass and abundance in stream communities in New Zealand (Scrimgeour et al. 1988; Quinn & Hickey 1990; Winterbourn 1997; Biggs et al. 1999). Floods can result in large sediment input or substrate scouring that could reduce the spawning success of some fishes or decrease production of some aquatic invertebrates. Major floods can cause substantial reductions in taxonomic richness, and total invertebrate biomass and density (Sagar 1986; Scrimgeour et al. 1988; Quinn & Hickey 1990). In general flood flows need to exceed c. 20 times the median flow to have significant effects on invertebrate abundance and taxonomic richness (Quinn & Hickey 1990). However, smaller more frequent events are capable of having significant influence (Clausen & Biggs 1997), supporting the importance of the frequency of flooding.

As stated in Section 5.2, habitat preference curves are meant to be obtained from unexploited streams at carrying capacity. Such conditions must be rare in high-disturbance New Zealand streams, and any habitat measurements taken during summer low flows (normal practise internationally and here) will have limited relevance.

As noted by Biggs (2000), a reduction in base flow velocity with abstraction or diversion of flow has the potential to greatly enhance the peak biomass of periphyton in enriched gravel- or cobble-bed streams. Phytoplankton and macrophyte growth is often enhanced in warm water at low flows (MFE 1992), particularly if the flow is relatively stable (Henriques 1987). Relatively high flows (around three times the median flow) are required to flush out this material (Clausen & Biggs 1997).

Sustained periods of low flows can lead to encroachment of weed species (e.g. crack willow *Salix fragilis*, broom *Cytisus scoparius* and gorse *Ulex europaeus*) onto riverbeds (e.g. Waitaki catchment: Maloney et al. 1999), thus reducing the areas of exposed gravel bed required by many terrestrial species for breeding and foraging. In addition, exotic weeds are thought to stabilise gravel bars, promote channel incision and decrease shallow-water foraging areas for these species and increase their risk of predation (citations in O'Donnell 2000). However, according to Hudson (2003), exceptional counts of key threatened bird species were related to the timing of channel clearing floods and subsequent steady low flows in the Ashburton River. There was no relationship between bird numbers and breeding period low flows.

Although the spawning and larval rearing habitat requirements of many New Zealand native fish are poorly known, some species are known to lay their eggs beneath stones (flathead galaxiids: Allibone & Townsend 1997; common river galaxiids: Cadwallader 1976) or on stream banks (inanga, banded kokopu: McDowall 1990) and would be equally vulnerable to siltation, or dehydration arising from prolonged lowering of water levels.

In contrast to the negative effects of flooding outlined above, floods at a watershed level may be critically important in refreshing stream systems, and fundamental to the maintenance of long-term productivity (Swanston 1991). Such floods can be responsible for creating habitats and increasing complexity by adding or redistributing gravel and wood, which may in turn increase stream productivity or provide refugia for fishes. Periodic flushing flows are also required to flush spawning redds (Milhous 1996) and feeding areas (Berkman & Rabeni 1987), with associated beneficial effects for stream fish communities (Lapointe & Payne 1996). Flushing flows may be required to induce spawning and egg hatch (banded kokopu, inanga: McDowall 1990).

Assessment of flow variability was the essence of an IFIM analysis from the onset, because it was recognised that minimum flows provide minimum protection (Stalnaker 1979; Stalnaker et al. 1995; Bovee et al. 1998). This applies not only to the active river channel, but also to the maintenance of the channel and riparian margin and receiving waters (Hudson 2002). Consideration must also be given to high flows as a population control. Bovee et al. (1994) found that population-related habitat limitations of bass in the Huron River, United States, were associated with high flows more often than with low flows (although both occurred). They noted that IFIM negotiations that focus only on minimum flows may preclude viable water management options and ignore significant biological events.

6.2.6 Lateral linkages

Overbank flows are required to develop and maintain riverine-riparian-floodplain-transitional upland fringe features and processes (see Section 2.3). Lateral movement of water into the floodplain during high flows, as well as contributing necessary water to the floodplain, puts the river system in contact with floodplain resources such as macroinvertebrates and microinvertebrates (Junk et al. 1989; Kellerhals & Church 1989; Sedell et al. 1989). As high flows decline, water returning from both terrestrial and aquatic systems on the floodplain brings food resources such as forest litter back to the mainstream (Junk et al. 1989; McGinness et al. 2002; Olley 2002). Floodplains, particularly their aquatic component, probably represent a strategic supply of biota, food organisms and habitat for riverine animals.

Consequently, the extension of the River Continuum Concept beyond the idea that a river's carbon load comes solely from upstream has been suggested. Pulsing of river discharge is considered to be critically important in maintaining lateral exchanges through river margins (Junk et al. 1989) and should be incorporated into approaches assessing the flow needs of a particular river (Hudson 2002; Thoms & Parsons 2002). Depending on the river, the interaction between floodplain and riparian areas, and land and river management practices, may be a key issue in sustaining river resources and in maintaining ecosystem integrity (Minshall et al. 1985; Junk et al. 1989; Kelly & Harwell 1990; Auble et al. 1994; Duel et al. 1994; McGinness et al. 2002).

Lateral and longitudinal linkages undoubtedly are important for native fish in New Zealand rivers (e.g. Collier 1995; Edwards & Huryn 1995; Harding & Winterbourn 1995; Townsend et al. 1997). Cyclical floods that inundate floodplains may provide access to or refresh critical habitats for fish species

(Junk et al. 1989). New Zealand species for which this is likely include inanga, banded kokopu, eels and giant kokopu, which are often found in stranded oxbows or flood channels (Taylor 1988; Chadderton 1990).

Terrestrial plant communities also require regular inundations. Fewer inundations and lower soil moisture levels may have implications for specialist riparian species that require soils that are seasonally waterlogged for seed bed preparation, distribution and watering of the seed (Rood & Mahoney 1993). Within New Zealand some riparian flood plain species (e.g. the threatened plants *Oleria polia*, *O. hectorii* and *Pittosporum obcordatum*) are known to favour sites variously affected by flooding and siltation; they may be maintained by periodic flood disturbance that reduces competition from taller trees (Clarkson & Clarkson 1994; Rogers 1996) and that promotes periodic establishment of new cohorts (e.g. *P. obcordatum*; Clarkson & Clarkson 1994).

Flood plain communities have been rarely considered in New Zealand when setting environmental flows, and certainly are not catered for by IFIM methodologies used in this country. However, the need to include these features has been explicitly recommended in recent IFIM practice: 'If the stream has a floodplain, it should be measured routinely as a functional part of the river channel' (Bovee 1997).

6.2.7 Longitudinal linkages

In addition to the upstream passage requirements, flows are required to maintain river mouth openings and provide favourable flow conditions for passage of diadromous species (see Section 6.1.2). Equally challenging for river managers is determining the inflow needs of bays and estuaries. Estuaries and their adjacent wetlands—coastal vegetative areas with inundated or saturated soils—serve as important spawning grounds for many species of fishes and support a variety of migratory birds (Sedell et al. 1989). These areas depend on inflow of fresh water from rivers and streams; tidal variance and estuarine structure are crucial for spawning and fish passage (McDowall 1988, 1990).

Abstraction of instream water resources may damage these ecological processes (Drinkwater & Frank 1994). Flow manipulations in rivers can significantly modify productivity (Yin et al. 1997), and morphology and hydrodynamics of the coastal zone (Kirk 1991). Negative changes to migration patterns, spawning habitat, species diversity and distribution and the production of lower trophic levels as well as fisheries have all been documented following severe hydraulic modifications (Drinkwater & Frank 1994). The collapse of the toheroa population in Te Waewae Bay (Carbines 1997) immediately followed the diversion of the Waiau River to Doubtful Sound. International studies (e.g. Yin et al. 1997) suggest the collapse is probably due to loss of sediment and/or nutrient flux from the river.

6.2.8 Channel evolution

In physical habitat simulation the channel structure (width, bed configuration and substrate composition) is assumed to be constant over the range of flows of concern (Petts & Maddock 1994). However, changes in channel morphology, in response to the changed flow and sediment transport regimes (resulting from changing flow regimes and/or reservoir trapping), alter the hydraulic

characteristics of the channel (depth, velocity and shear stress distribution), substrate characteristics (Petts 1979; Williams & Wolman 1984) and space and shelter availability (Petts 1989). For example, because of sediment trapping behind dams, downstream scour often occurs (Williams & Wolman 1984). Channel width changes also occurred, with a tendency to narrower widths with smaller flows and vegetation encroachment. Such changes will have a major impact upon both benthic invertebrates and fish (Welcomme 1979; Milhous 1982).

Deposition in regulated rivers may occur because peak flows are reduced but the sediment supply continues from tributary streams. For example, in the Colorado River below Glen Canyon Dam, Arizona, deposition occurred at the confluence of every major tributary (Howard & Dolan 1981). The bed may trap fine sediment thus cementing silt in the bed (Reid et al. 1997; ASACE 2002), or fine material may be removed from the bed (Petts 1984). In addition, channel changes can alter the nature of river-floodplain interactions, and isolate the main river from its river plain, eliminating access to backwaters, floodplain lakes and marshes (Petts 1989).

There are two types of issue related to channel dynamics and stability (Bovee et al. 1998): determining flow requirements to prevent the channel from changing; and predicting how channel changes are likely to affect instream habitat. Understanding the influence of changing flow regimes on habitat structure remains a neglected research area in stream ecology (Hill et al. 1991; Biggs et al. 2001; Power 2001) and the techniques available to predict channel responses to changes in flow regime or sediment transport are crude (Bovee et al. 1998). A non-changing bed has almost invariably been assumed in IFIM studies.

6.3 MICROHABITAT INTERACTIONS

The assumption of independence of variables—that depth and velocity preferences determined at one flow predict fish distributions at another flow (Section 5.3.1)—has been questioned from the onset of IFIM. Gore & Judy (1981), Orth & Maughan (1982), Mathur et al. (1985), Morin et al. 1986), Orth (1987), Scott & Shirvell (1987) and Gore & Nestler (1988) suggested that optimum values and ranges of HSC values change with changing physical environment (and with the age structure of the population). For example, Shirvell (1990) found that juvenile steelhead used areas of stream with different water velocities at different flows, which suggests that the microhabitat preferences of juvenile steelhead may be a function of flow. Pert & Erman (1994) and Shirvell (1994) found that adult rainbow trout and juvenile coho and chinook salmon moved to new positions within a stream after changes in streamflow, i.e. their new locations were characterised by a new set of microhabitat preferences. Shirvell (1994) also found that when the primary purpose was feeding, fish selected optimal positions along velocity gradients as suggested by Fausch (1984). However, when the primary purpose was avoiding predators, fish selected optimal positions along light gradients as found by Heggenes et al. (1993). Dedual (2000) found rainbow trout fry depth and

velocity preferences changed in the Hinemaiaia River, North Island, as flow changed.

These results have important implications for habitat simulation, at least for salmonids. Habitat use and preference curves collected at one narrow range of discharges may be inappropriate for assessing potential fish responses at other discharges, and they may affect predictions of the effects of changes in streamflow on fish habitat (Pert & Erman 1994; Shirvell 1994).

This situation is less clear for native fish in New Zealand. A study by Jowett & Richardson (1994) examined habitat use in normal and flood conditions on the Pohangina River, North Island, for four depth-stratified lanes below 0.75 m deep. Data on smelt, shortfin eels, Cran's bully and upland bully indicated that more work is required but it seems likely habitat variables are probably not independent in their effect on fish position choice. This dependence leads to a skew of fish distributions within a stream (Scott & Shirvell 1987). Alternative approaches to HSC development are required (see Sections 3.1 and 5.3.2).

7. Verification and validation

Verification and validation are considered essential components of an IFIM analysis (Table 1) (Stalnaker et al. 1995). 'Backcasting' (Bovee 1982) is one technique: it is a historical analysis using effective habitat time series analysis designed to evaluate past habitats. Historical records of fish standing crop are used to back-calculate year-class strength and growth histories. Potential historic habitat limitations can be identified (e.g. extreme low flows during droughts) and can then be examined in terms of indicators of population response (e.g. year-class strength, growth rates, adult survival). Good habitat years and poor habitat years can be used to 'calibrate' the habitat time series and establish validity for the model simulations.

Field verification is the official policy of the US Fish and Wildlife Service because of concerns related to the transferability of habitat-use criteria (see Section 5.1). Unfortunately, many, if not most, past applications of IFIM have not involved collecting sufficient data to verify that the models had behaved as intended or that assumptions (e.g. that fish populations respond to habitat limiting events) were valid (Stalnaker 1994, Stalnaker et al. 1995).

Validation is also required to address the basic statistical questions about sampling procedures and HSC development (see Sections 4.3 and 4.4). Model results need to be compared with independent, randomly collected data, as is the ordinary practice in science, but no PHABSIM study seems to have been verified in this way (Williams et al. 1997). Theoretically, this remarkable lapse from scientific practice is by itself grounds for dismissing the study results (John G. Williams, consultant, Davis, California, pers. comm.).

8. Training

Criticisms of IFIM analysis from a sampling theory viewpoint have included concerns raised by users of the methodology in western North America (Armour & Taylor 1991). These authors reported that most users of the methodology felt that some level of training of field workers was essential to standardise different levels of experience for sampling different streams and different reaches of the same stream (Armour & Taylor 1991). Wang et al. (1996) concurred and also emphasised that if transect measurements of physical habitat characteristics in streams are used, initial field training must be adequate to achieve reasonable accuracy, precision and repeatability of measurements among streams, and even between different reaches of the same stream.

Inexperience and inadequate training of field researchers can also lead to uncertainty in measurement of cross-sectional transects and the frequency of measurements along a transect (Shirvell 1989; Simonson et al. 1994), both of which can have important consequences for the way the stream is sampled (Shirvell 1989). Problems in the identification of habitat units, and in the description of substrate and cover, were discussed in Section 4.

9. Resource Management Act considerations

In New Zealand, the Resource Management Act 1991 (RMA) establishes a broad environmental ethos within which management of natural resources (including rivers) is to focus on sustainability, in terms of ecological functions, intrinsic values and potential value for future generations. To date, however, there has been little discussion of what the RMA actually means in practical, flow-related terms (Day & Hudson 2001). Managers have often focused exclusively on low flows and their ability to provide adequate habitat for recreational and commercial fisheries (see Section 2). Landscape, recreational and Maori traditional uses of New Zealand waterways are sometimes acknowledged, yet with the possible exception of a few major consents, these alternative needs seem to rarely have had much influence on setting and maintaining flow requirements in New Zealand.

Currently, regulated stream flows sometimes define how much water can be released from a dam and how much water should be left in a river after abstraction. Minimum specified flows, often determined by rules or regulations, are used by power companies and local bodies to maintain flows at low levels for extended periods, and there is some concern that the low flow value may become a maximum as well as a minimum flow. Should this happen, it would be difficult, legally, to re-negotiate environmental flow requirements.

Many New Zealand IFIM analyses have focused on setting a minimum flow to maintain water depths and velocities for a few target species in selected river reaches. Arbitrary 'acceptable losses' of habitat (in the Wellington region, a one-third loss) have been suggested (Jowett 1993b). These limitations are counter to best international practice and the spirit and intent of the Resource Management Act.

Internationally it has been demonstrated that the magnitude, frequency, duration, timing and the rate of change in flow are important components of river ecosystems (Poff et al. 1997). In New Zealand there has been little acknowledgement of the need for flushing flows for spawning and feeding of native species as well as for introduced salmonids, or for the periodic flooding of riparian, floodplain and estuarine areas, particularly with regard to their effects on New Zealand native fishes (McDowall 1990, 1995).

We believe that in New Zealand there is also a need to broaden our focus away from the concept of a single minimum flow and move to the environmental flow concept (see Section 2). That is, not only do we need to broaden the spatial focus of IFIM assessments, but also to broaden the target species. Given the lack of detailed understanding of many of the life history requirements of New Zealand's native fish species, it is poor management practice to assume that minimum flow guidelines established for salmonids will necessarily be adequate to maintain ecologically viable populations of native fish species.

In terms of interpreting WUA-discharge relations, and determining an appropriate level of ecosystem protection, the MFE (1998) flow guidelines state flows can be set so that they:

1. Maintain optimum levels of fish habitat;
2. Retain a percentage of habitat at average or median flow; or
3. Provide a minimum amount of habitat.
4. Flows can also be set at the point of inflection in the habitat/flow relationship. This is possibly the most common method of assessing minimum flow requirements using habitat methods. While there is no percentage or absolute value associated with this level of protection, it is a point of diminishing return where proportionally more habitat is lost with decreasing flow than is gained with increasing flow.'

As mentioned earlier, maintaining optimum habitat levels is a requirement in some jurisdictions (Canada, Australia, South Africa and parts of the United States; see Section 2.3). Even providing a minimum amount of habitat implies considerable ecosystem knowledge. We do not know the basic habitat requirements of many of our native species and much is still being learned regarding well known species such as brown trout which may change how we describe their requirements in habitat simulation (e.g. Hayes et al. in press).

A current tool, 'inflection points' (point 4, above), can be unreliable (Gippel & Stewardson 1998) and imply both a hydro-morphological and biological meaning to the shape of WUA-flow relations that should be demonstrated with each use. Another tool, mesoHABSIM analysis (Parasiewicz 2001), documents the variation in the spatial distribution and amount of mesohabitat with changing streamflow, and the mesohabitat changes can then be related back to inflection points in WUA-flow relations (e.g. dewatering of seep channels and braids: Hudson 2001c, 2003).

Maintaining environmental flows is considered to be an acceptable cost to society. Examples include capping water takes on the Murray River, Australia, at 1993/1994 levels, with a view to progressive reductions (Rogers et al. 1997); reductions in total water supply for urban and agricultural use in California (California Water Commission 1994); and compensation flows in the Snake and Columbia Rivers system (worth \$350 US million per year) (Prendergast 1994). Maclin & Sicchio (1999) have described over 50 dams in the United States that have been removed primarily for ecological reasons.

Environmentally sustainable flow regimes are not necessarily incompatible with the needs of other water users. The objective of the flow negotiating process is to rigorously determine the instream flow requirements for environmental and other instream uses, and to balance the environmental needs with the social and economic needs of the community. This, in our opinion, is the essence of IFIM and of the Resource Management Act.

10. Summary and recommendations

By our paper we wish to encourage a re-evaluation of the use of the instream flow incremental methodology (IFIM) in New Zealand, particularly the use of habitat simulation models. We were concerned that habitat simulation (specifically RHYHABSIM, the New Zealand variant of PHABSIM) was being promoted as '**the tool**' for recommending minimum flows in streams, without consideration of limitations in the methodology, limitations in how the modelling is applied or limitations in perspectives.

Since the development of IFIM there has been continued significant criticism of the biological, physical and methodological basis of its associated physical habitat simulation (e.g. Mathur et al. 1985; Scott & Shirvell 1987; Shirvell 1989, 1994; Bourgeois et al. 1996; Heggenes 1996; Williams 1996; Kondolf et al. 2000). The IFIM developers have acknowledged some of the criticisms: '... there are many interactions between species, life stages, and other variables that influence the state of the ecosystem that are not modelled by PHABSIM [and] ... the physical habitat is a necessary but not sufficient condition for a viable population of aquatic animals ...' (Milhous 1999a). However, in our opinion, limitations and criticisms have been essentially treated as truisms and ignored.

Leading experts in instream flows, in response to a court-ordered review, concluded: '... there is now no scientifically defensible method for defining flow standards ...' and 'We have divergent views on PHABSIM. Some of us think that, with modification and careful use, it might produce useful information. Others think it should simply be abandoned' (Castleberry et al. 1996). This panel outlined how PHABSIM (or its variants) should be used, indicating a variety of problems that should be addressed (e.g. sampling and measurement problems describing rivers, see Sections 4 and 5.2; sampling and measurement

problems associated with HSC, see Section 5; and problems with assigning biological meaning to WUA, see Section 5.3.3) (Castleberry et al. 1996). We address these below.

In addition, there are issues in the mechanics of undertaking a habitat simulation analysis.

10.1 PROBLEMS DESCRIBING RIVERS

For particular reaches of a river, a wide divergence in outcomes is possible because of sampling problems (e.g. Wang et al. 1996; Williams 1996; Dollof et al. 1997; Hudson 2001a), the methods of calculation within PHABSIM for a given reach (Gan & McMahon 1990), hydraulics modelling problems (e.g. Kondolf et al. 2000), choice of habitat suitability curve for a particular species and life stage (e.g. Glozier et al. 1997) and weight given to particular species and life stages in recommending a flow regime.

One aspect of sampling is transect placement. Often complex reaches, or difficult reaches (e.g. too deep to wade, to shallow to jetboat), are avoided. Transects placed for habitat mapping can also be problematic, because major habitat units are often improperly defined both in field surveys and in the retrospective delineation of habitat types using a hydraulic criterion (e.g. Jowett 1993; Roper & Scarnecchia 1995; Hudson 2001a). Approaches are available to rigorously define habitat variability between river segments and to determine appropriate sampling locations within these segments (e.g. Hardy & Addley 2001).

In terms of substrate, Kondolf (2000) stated there is no evidence that visual estimates of substrate size classes, as normally undertaken for PHABSIM, are reproducible between investigators. In fact, in one New Zealand river, an investigator reported contradictory findings regarding substrate variability in two separate studies (Mosley 1983; Mosley & Tindale 1983).

At a reach scale, there is an inability to model complex hydraulics in PHABSIM and equivalent 1D models (Kondolf et al. 2000). Waddle et al. (1997) suggested that 2D models may provide a means to overcome some of these difficulties, although appropriate river reaches must still be selected, and appropriate sites surveyed at appropriate scales.

Gan & Mahon (1990) demonstrated that many options can be exercised both between and within programs in the PHABSIM suite, such that a wide variety of results can be obtained from a single set of data. There are fewer options in RHYHABSIM (the variant used in New Zealand), which increases repeatability but probably at the expense of accuracy.

Assuming PHABSIM procedures are correctly followed and sampling transects are perfectly placed, uncertainties in estimates of WUA may be great. In one study, a five-fold difference in the flow that would yield peak WUA was estimated for a river in California, for a given species and life stage (Williams 1996). To aid interpretation of WUA estimates, they should always be presented with confidence intervals (Castleberry et al. 1996).

Changing hydro-geomorphic characteristics profoundly influences the WUA-streamflow relationship (e.g. within and between braided and single channel reaches: Beschta & Platts 1986; Heede & Rinne 1990; Rabeni & Jacobson 1993; Payne & Lapointe 1997; Hudson 2001c; Table 2). Correctly describing these changes is an inference problem in which information obtained from a relatively small sample is extrapolated to a population (a length of river that varies both in time and in space). There is no theoretical justification for subjectively chosen sampling sites. According to Williams (1996), there is often no valid statistical basis for extrapolating from such survey results.

There is also the problem of developing appropriate metrics, at appropriate scales, for describing habitat attributes of ecological importance (e.g. Bult et al. 1998; Kocik & Ferreri 1998; Crowder & Diplas 2000; Kondolf et al. 2000). In hydraulic modelling transects are placed tens to hundreds or more metres apart in 1D modelling and the survey grids for 2D models have spacings of metres to tens of metre. The actual area of use by fish and other aquatic animals is often in the order of a metre (Grant & Kramer 1990; Hill & Grossman 1993; Hughes 1998). Flexible mesh models (e.g. Rivers2D: Blackburn & Steffler 2002) provide the flexibility to model complex structures (e.g. riffles, rootwads) at an appropriate scale. Fixed grid models (e.g. Duncan & Hicks 2001) can not effectively model these complex features.

The complex relations between structure, microhabitat and use are also problematic. Flexible survey and computational grids can be used to describe habitat variability in 2D hydraulic modelling. However, as noted by Freeman et al. (1997) and Stewart (2000), habitat structure may more strongly influence habitat suitability for many stream animals (e.g. salmonids, small mouth bass, and many cyprinids) than does the occurrence of a particular range of microhabitat conditions used in PHABSIM (i.e. depth, velocity, substrate). A MesoHABITAT approach, in which changes to habitat structures are assessed, is probably more meaningful (Parasiewicz 2001).

10.2 PROBLEMS WITH HABITAT SUITABILITY CURVES

Habitat suitability curves, although critically important (Gore & Nestler 1988), remain one of the most controversial aspects of PHABSIM (e.g. Armour & Taylor 1991; Glozier et al. 1997). There are significant problems in translating observations into biologically meaningful habitat suitability curves (e.g. Guay et al. 2000; Hayes et al. in press).

Currently there are large gaps in our knowledge of detailed life histories and habitat requirements of New Zealand fish species, particularly galaxiids. This applies to instream habitat requirements and behavioural characteristics, as well as estuarine and ocean requirements of New Zealand's diadromous fishes. Widely used habitat suitability curves were developed from a limited range of environments and from a narrow range of conditions, but the indices have been applied generally, without testing (e.g. Chadderton & Allibone 2000; Bonnett & Sykes 2002).

Although validation of habitat suitability curves is required by agencies such as the United States Fish and Wildlife Service, reviews have shown that there are few studies that have done so (Armour & Taylor 1991; Thomas & Bovee 1993; Stalnaker et al. 1995). More procedures to develop HSC (five in Bovee et al. 1998), and to test transferability (Thomas & Bovee 1993) have been developed. However, different methods of deriving habitat suitability curves produce different results (e.g. Beecher 1995) and Glozier et al. (1997) could not recommend the use of Thomas & Bovee's (1993) transferability tests as a definitive answer on HSC transferability. Glozier et al. (1997) supported the suggestions from previous researchers that HSC are best generated on a system specific basis.

Glozier et al. (1997) and Freeman et al. (1997) suggested that other variables (in addition to depth, velocity and substrate) determine fish distribution and a study of chinook salmon supported this (Geist & Dauble 1998). It would seem that conventional habitat suitability curves do not adequately describe critical flow-dependent features of a habitat. In addition, critical requirements such as groundwater upwelling in salmon spawning areas are known but not modelled resulting in unrealistically high estimates of habitat availability and use (Shirvell 1989; Geist & Dauble 1998).

Promising alternative approaches to develop HSC have been proposed (Braaten et al. 1997; Guay et al. 2000; Hayes et al. 2003).

10.3 PROBLEMS WITH BIOLOGICAL MEANING OF WUA

The index of habitat availability that PHABSIM generates—WUA in m^2/m —is difficult to imbue with biological meaning (Orth 1987; Heggenes 1996). It provides no meaning as an indicator of where fish are located. One advantage of 2D modelling is that spatial patterns can be discerned (e.g. Waddle et al. 1997).

Most studies are based on the assumption that there is a positive relationship between fish biomass and WUA, however this relationship (in terms of fish abundance) should be demonstrated before the IFIM-PHABSIM method can be considered valid for assessment of instream flow requirements (Jowett 1992). Positive relationships have been found between fish abundance and WUA in some streams in the United States (e.g. Conder & Annear 1987; Milhous 1999a) and between brown trout WUA and food WUA in New Zealand (Jowett 1992), but in the latter many other variables are required to explain fish abundance (see footnote 3). There are many more observations of poor or negative correlations (e.g. Orth & Maughan 1982; Irvine et al. 1987; Pert & Erman 1994; Zorn & Seelbach 1995; Bourgeois et al. 1996). For example, it is estimated that there is virtually no available habitat for spawning or young fish at flows that occur more than 75% of the time in the lower Tongariro River. In contrast, Jowett et al. (1996) report spawning in the De Latours reach and relatively high densities of young fish below the SH1 bridge.

10.4 FROM MINIMUM FLOWS TO ENVIRONMENTAL FLOWS

In New Zealand IFIM studies, a narrow perspective of streamflow requirements has usually been taken: the focus has often been on minimum flow (survival flows) for target fish species without consideration of critical elements such as flow variability, and maintenance of ecosystem processes. Stream temperature and water quality are two overriding constraints on habitat availability, both depend on flow conditions and yet neither are normally rigorously considered in setting streamflow regimes in New Zealand.

Rather than minimum flows, environmental flows should be the focus. Environmental flows provide a flow regime for the river corridor (i.e. the channel itself as well as the floodplain, and the transitional upland fringe) and receiving waters (e.g. coastal zone), for the purpose of maintaining ecosystem structure (e.g. wetlands, oxbow lakes) and processes (e.g. nutrient cycling; sediment flux) in their own right (Hudson 2002). To satisfy the spirit and intent of the Resource Management Act, environmental flows, not minimum flows, are required (Day & Hudson 2001).

10.5 SUGGESTED APPROACH

An ecosystem approach to flow regime management involves at least the following steps (Hudson 2002).

- Determine the physical nature of the entire riverine ecosystem at multiple scales (river segments of similar discharge and sediment regimes; river reaches; habitat structures and communities; and microhabitats).
- Identify the significant ecological requirements and processes associated with the significant physical structures and flow characteristics in the river corridor (in-channel flow; riparian flows, floodplain and upland fringe forming flows).
- Identify the significant ecological requirements and processes associated with the significant physical structures and flow characteristics in receiving waters (e.g. other rivers, deltas and nearshore zone of lakes and the ocean).
- Identify the key hydro-geomorphological drivers, and the implications of change (e.g. impoundments reducing flow and sediment) on physical habitat structure and processes.
- Derive key management goals and objectives for each of the significant structures and processes for each segment of the river.
- Evaluate and implement management options.
- Critically evaluate outcomes and enhance the science, goals and policies.

There are several major challenges in implementing these steps (Hudson 2002; Thoms & Parsons 2002). The expert panel (e.g. Thoms & Swirepik 1998) and building block methodologies (e.g. King & Louw 1998; Arthington 1998) provide guidance.

Regarding the last point above, modelling results are often taken as gospel, in spite of expert knowledge and known limitations. To quote Platts (1981):

'Stream inventory garbage in—reliable analysis out: only in fairy tales.' This is recognised by Castleberry et al. (1996) who cautioned that an analytic method should not become a substitute for common sense, critical thinking about stream ecology or careful evaluation of the consequences of flow modification.

It would be prudent to treat flow manipulations as experiments. Management objectives and expectations must be explicitly stated and based on the best available knowledge; management approaches must be viewed as hypotheses to be tested by research and monitoring; management must also be able to adapt to new information and understanding; and public understanding and involvement are required (Hudson 2002).

To make informed decisions, managers require measures of confidence on scientific recommendations (e.g. flow regimes). They need to accept the existence of uncertainty and make decisions that will both protect resources and allow development of knowledge that will reduce the uncertainty (Castleberry et al. 1996). Comprehensive monitoring and assessment is required, and both success and failure of habitat improvements, and flow regime recommendations, must be reported, in order to learn and provide better advice (Hudson 2002).

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Appendix 1

COMMON AND SCIENTIFIC NAMES OF ANIMALS MENTIONED

SCIENTIFIC NAME	COMMON NAME
Fish	
<i>Anguilla australis</i>	Shortfin eel
<i>Anguilla dieffenbachii</i>	Longfin eel
<i>Cheimarrichthys fosteri</i>	Torrentfish
<i>Galaxias anomalus</i>	Roundhead galaxias
<i>Galaxias argenteus</i>	Giant kokopu
<i>Galaxias brevipinnis</i>	Koaro
<i>Galaxias fasciatus</i>	Banded kokopu
<i>Galaxias maculatus</i>	Inanga
<i>Galaxias postvectis</i>	Short jawed kokopu
<i>Galaxias prognathus</i>	Longjaw galaxias
<i>Galaxias vulgaris</i>	Common (river) galaxias
<i>Gobiomorphus cotidianus</i>	Common bully
<i>Gobiomorphus breviceps</i>	Upland bully
<i>Gobiomorphus buttoni</i>	Redfinned bully
<i>Gobiomorphus hubbsi</i>	Bluegilled bully
<i>Micropterus dolomieu</i>	Smallmouth bass
<i>Oncorhynchus kisutch</i>	Coho salmon
<i>Oncorhynchus mykiss</i>	Rainbow troutSteelhead trout
<i>Oncorhynchus tshawytscha</i>	Chinook salmon
<i>Retropinna retropinna</i>	Common smelt
<i>Salmo trutta</i>	Brown trout
<i>Salmo salar</i>	Atlantic salmon
<i>Salvelinus fontinalis</i>	Brook trout
Invertebrates	
<i>Pappies ventricosa</i>	Toheroa