A guide to instream habitat survey methods and analysis

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Contents

Abstract

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Although instream habitat analysis is widely used in New Zealand and globally, its use has not been without controversy. The purpose of this guide is to address the concerns about the method, and to establish a creditable base for the continued use of the method and future developments. The guide is not intended to be a basic manual for instream habitat analysis.

The concept of habitat suitability is familiar to most people who collect fish, plants, or aquatic insects from rivers. Most authoritative descriptions of aquatic biota include qualitative descriptions of the habitats and physical conditions in which the biota are likely to be found. In the aquatic environment, instream habitat usually refers to the physical habitat (water velocity, depth, substrate, and perhaps cover), and can include the physical characteristics of larger habitat units, such as pools, runs, and riffles. The quality of the different habitats in streams is defined by the relative abundance of animals in them. Usually, animals will be most abundant where the habitat quality is best, in lesser numbers where the habitat is poor, and absent from unsuitable habitat. Quantitative relationships between physical habitat and species abundance or presence/absence are used to construct habitat suitability models or curves. Instream habitat methods use these curves with hydraulic models to determine the distribution of suitable habitat, and how the amount of suitable habitat or weighted usable area (WUA) varies with flow.

The derivation and use of habitat suitability models are the most important aspects of flow assessment. Although habitat suitability criteria are available for many New Zealand aquatic organisms, they can be improved by collecting more data and recalculated habitat suitability models.

Instream habitat methods, although often described as evaluating microhabitat, are in fact evaluating mesohabitat. The survey techniques described here are capable of predicting depths and velocities to the scale of the survey, which usually has measurements spaced at 0.1–3.0 m. They do not predict micro-scale hydraulics. Similarly, many habitat suitability observations describe mesohabitats – the characteristics of the area in which the organism lives, rather than the micro hydraulics of its precise location. In assessing suitability for one species, we are often assessing conditions for a number of species that live in that area. Riffle-dwelling fish and invertebrates are an example, where the habitat suitability curves describe riffle conditions, rather than microhabitat of the location of an individual organism. The selection of an appropriate critical values or target species (fish or invertebrate) as an indicator of stream health is a concept that can be applied to flow assessment.

Environmental flow assessments usually consider WUA-flow relationships for a range of aquatic species and decisions as to an appropriate minimum flow are based on maintaining habitat for target values (often those with the highest flow requirement). The main environmental concern about minor water abstractions will usually be the minimum flow. However, large-scale projects like damming and major diversions will usually require detailed and specific studies to determine downstream flow regime requirements, including the duration and frequency of low flows, flushing flows, and channel maintenance flows. Although the functional role of these flow regime components is known, we do not know the degree to which the frequency and duration of these events affect biota and do not have any quantitative method of assigning acceptable frequencies and durations, other than mimicking nature. However, for benthic invertebrates and periphyton it is possible to provide rough guidance on the frequency of flushing flows, based on a conceptual understanding of colonisation processes and reported colonisation times.

The examples presented in this guide show that rivers are hydraulically similar, i.e., they have similar shaped habitat/flow relationships, over long sections of river. The selection of reaches is neither a

complicated nor sensitive task and the variation of habitat with flow can be determined from relatively few cross-sections. In fact, habitat analyses based on simple hydraulic geometry, 1D surveys or 2D surveys, will produce useful and similar results. However, the tasks of survey, calibration, habitat suitability analysis, and finally the interpretation of results, require a good knowledge of river mechanics, hydraulics, and ecology. The survey (habitat mapping) and hydraulic calibration procedures used in RHYHABSIM are relatively robust, but more complex modelling techniques, such as water surface modelling and 2D modelling, are more difficult and can be done poorly, both in the execution of the survey and hydraulic analysis.

Finally, hydraulic habitat modelling is a tool to assist the decision-making process. No flow will maintain maximum habitat for all aquatic organisms, because different organisms have different instream habitat requirements. The selection of an appropriate flow regime for a river requires clear goals and target objectives, with levels of protection set according to the relative values of the in- and out-of-stream resources. The process of establishing target objectives must be realistic and focused – objectives should be relevant, important, flow dependent, and hierarchical. Failure to establish clear management goals and to carry out wide consultation will lead to conflict. Attempts to maintain everything in the existing state invariably lead to the conclusion that flows should not be changed, and precludes the opportunity for enhancement of some aspects of the aquatic environment and use of the water resource.

1. Introduction

Organisations responsible for water management are becoming increasingly aware of their responsibilities for environmental protection, creating an increasing interest in methods of assessing flow requirements for different instream uses. In Europe, there are attempts to rehabilitate large rivers that have been controlled and channelised for centuries. In the United States, attempts are being made to rehabilitate the lower Mississippi River and, in Australia, the extensive flow regulation of the Murray-Darling River system is being questioned. Water managers in New Zealand, although operating on a smaller scale, are required to assess the impact of water use on the stream environment whenever development of the water resource is proposed, or when the rights of use for that resource are reviewed.

Tharme (2003) reviewed international trends in environmental flow management and divided flow assessment methods at two levels: (i) reconnaissance-level initiatives relying on hydrological methodologies formed the largest group (30% of the global total), where commonly a modified Tennant method or arbitrary low flow index is adopted, and (ii) a more comprehensive scale of assessment where the instream flow incremental methodology (IFIM) or other similarly structured approaches are used (28% of global total).

According to a review by the Environment Agency in the UK on river flow objectives, ‘internationally, an IFIM-type approach is considered the most defensible method in existence’ (Dunbar et al. 1998). The Freshwater Research Institute of the University of Cape Town (Tharme 1996) states: ‘IFIM is currently considered to be the most sophisticated, and scientifically and legally defensible, methodology available for quantitatively assessing the instream flow requirements of rivers’. A review of flow assessment methods (Annear et al. 2002) described IFIM as the ‘most appropriate for relative comparisons of habitat potential from among several alternative flow management proposals’, and as ‘the method of choice when a stream is subject to significant regulation and the resource management objective is to protect the existing healthy instream resources by prescribing conditions necessary for no net loss of physical habitat’. Nevertheless controversy has accompanied the development of the IFIM, in particular hydraulic and habitat models such as RHYHABSIM and PHABSIM (e.g., Mathur et al. 1985; Scott & Shirvell 1987; Kondolf et al. 2000; Hudson et al. 2003). Despite these repeated criticisms, IFIM and similar hydraulic habitat methods have a biological basis and are used in approximately 58 countries (Tharme 2003).

Tharme (2003) found that holistic methodologies formed 8% of the global total of environmental flow assessments and are an approach to minimum flow assessment currently favoured in Australia and South Africa. The aim is to maintain a natural flow regime and its low flows, seasonal variation, and flood frequency in order to protect aquatic fauna. A flow regime policy that restricts abstractions to the level of naturally occurring low flows and maintains major elements of the natural flow regime will maintain stream fauna, essentially in a natural state. This is a “safe” environmental policy and one that will ensure the protection of aquatic resources in most situations. A similar ‘standard setting’ approach has been outlined by Mathews and Richter (2007) who use the Range of Variability Approach (RVA) and associated Indicators of Hydrologic Alteration (IHA) to derive a recommended flow regime. The IHA characterises the flow regime using a large number of hydrological parameters. The IHA/RVA method has been used the USA mainly in regulated systems to maximise the benefit of high-flow pulse releases of water from dams at a targeted magnitude, frequency, timing, duration and rate-of-change. This method has not been used in New Zealand and the ecological relevance of the hydrological parameters has not been established (Tharme 2003).

This report covers the following aspects of instream flow assessments:

- the effect of flow regime on stream ecology, as a background to flow assessment methods
- methods of flow assessment that have been suggested and their effectiveness in achieving ecological objectives, and resolving the trade-offs between water-use and protection of instream values
- instream values, and the process of selecting appropriate environmental goals and levels of protection
- a detailed description of the application of physical habitat modelling
- case studies, where flow regime requirements have been based on instream habitat modelling and ecological responses monitored.

2. Stream ecology

The current is the driving force of a stream. It is necessary for the respiration of many benthic invertebrates and reproduction of some fish species (Hynes 1970; Aadland 1993; Moir et al. 1998; Gore et al. 2001; Mérigoux and Dolédec 2004). Currents distribute nutrients and food down a river system, detritus for invertebrates and drifting insects for fish, and aid species dispersal – part of the river continuum theory. Biologists and anglers who study rivers are well aware that aquatic species are likely to be found in association with specific habitats, and many aquatic species are found in similar hydraulic conditions in a wide range of rivers. These have been termed habitat niches, and include both physical and biotic characteristics of the environment (Odum 1971). The habitat niche concept is essentially the same as habitat suitability. These concepts have been widely applied in both terrestrial and aquatic biological studies, with the understanding that the presence of suitable habitat for any species is a necessary condition for survival.

Aquatic life in streams and rivers has developed under a ‘natural’ flow regime. If the instream environment under natural flows is unsuitable for a particular species then that species will not be well established in a stream. Periodic disturbances, such as floods and droughts, affect stream biota. Floods can reduce trout stocks (Jowett & Richardson 1989), invertebrates (Quinn & Hickey 1990), and periphyton (Biggs et al. 1999). However, the effect of disturbance frequency differs between aquatic species. If disturbances are too frequent and severe, biota will be unable to establish self-sustaining populations. New Zealand native fish and adult brown trout seem to be particularly well adapted to surviving large floods, even taking advantage of the situation to feed (Jowett & Richardson 1994). Aquatic insects are also relatively robust, generally recolonising a stream within months of a severe disturbance. Stream insects recolonise streams relatively quickly, either by drifting in from upstream, from refugia within the gravels, or from eggs laid by the adult insects following winged dispersal. The recolonisation rate of fish is slower than that of stream insects. Juvenile trout and adult rainbow trout are particularly affected by floods, presumably because they do not utilise cover as well as adult brown trout, and because juvenile fish are weaker swimmers.

The biota present in a stream have survived a series of historical disturbances and, presumably, will continue to survive, provided that the frequency and magnitude of these disturbances does not change. Some stream ecologists hypothesise that stream biota have adapted to the flow regime of particular streams or rivers, and in particular, they believe that biota have adapted to survive the low flows that occur in the river with reasonable frequency.

The mean annual minimum flow (MALF) is the average of the annual minimum flows. The minimum flow can either be the instantaneous, daily, or seven-day minimum flow; the advantage of the former being its ease of calculation; the advantage of the latter being that ‘spikes’ in the hydrological record have less influence on its value. Biologically, the mean annual minimum flow may be a ‘bottleneck’ for aquatic species that have life cycles in the order of three to five years. If low flows are a ‘bottleneck’, a reduction in minimum flow would have a detrimental effect, but if the species is not limited by low flows, then a reduction in minimum flow will have no effect.

The MALF is indicative of the low flows likely to be experienced during the generation cycles of trout. In small to medium sized rivers, low flows generally set the lower limit to physical space likely to be experienced by trout, although the duration of low flow may also be relevant.

The MALF is also similarly relevant to native fish species with generation cycles longer than one year in small rivers or streams where the amount of suitable habitat declines at flows less than MALF. Research in the Waipara River, where native fish habitat is limited at low flow, showed that the detrimental effect on fish numbers increased with the magnitude and duration of low flow (Section 11.8; Jowett et al. 2005). Research on the Onekaka River in Golden Bay also showed that, when habitat availability was reduced by flow reduction, abundance of native fish species responded in accord with changes in habitat availability in both direction and magnitude (Section 11.7; Richardson & Jowett 2006).

In contrast to long-lived fish species, some aquatic invertebrates have more than one cohort per year, and in New Zealand generally have asynchronous lifecycles (i.e., a range of different life stages are likely to be present at any given time), allowing them to rapidly repopulate areas following disturbance by drift from tributaries and from other rivers by winged dispersal. Recolonisation of some river beds by benthic invertebrates following disturbance has been reported to occur within four to ten weeks (Sagar 1983; Scrimgeour et al. 1988). Because benthic invertebrates can respond relatively quickly to available habitat conditions, the median or modal flow provides an approximation of the habitat conditions experienced, and able to be utilised, by benthic invertebrates most of the time.

2.1 Habitat requirements and relationships with abundance of aquatic fauna

Most aquatic insect and native fish species are found in a wide range of rivers and streams, from large to very small. Studies have been carried out to determine habitat preferences of native fish (e.g., Jowett & Richardson 1995), and these have been verified by other studies, which demonstrated that native fish are more abundant where the average stream characteristics are close to the preferred habitat for the fish species (Jowett et al. 1996a; Jowett 2006). Although some studies have shown small changes in galaxiid habitat use in the presence of trout (see review in McDowall 2006), the preferred habitats of most native fish species in areas, such as Northland where there are no or very few trout, are similar to those in streams further south where there are trout present (Richardson & Jowett 1998). Native fish densities are often higher in small streams than larger streams or rivers because the preferred habitat of many native fish species is relatively shallow water. Similarly, stream insects are often occur at higher densities in small streams than in larger streams and rivers.

It is the quality of the habitat that is provided by the flow that is important to density of stream biota, not the magnitude of the flow *per se*, although habitat quantity is also relevant when the aim is to maintain the maximum biomass of river ecosystems to sustain instream values such as fisheries. In many streams, flows less than the naturally occurring low flow are able to provide good quality habitat and sustain stream ecosystems. The magnitude of this flow will vary with the requirements of the species and with the morphology of the stream. Water velocity is probably the most important characteristic of a stream. Without it, the stream becomes a lake or pond. In gravel-bed rivers, an average velocity of 0.2–0.3 m/s tends to provide for most stream life, because velocities lower than this provide unsuitable habitat for a number of fish species and stream insects, and allow deposition of sand and finer materials, as well as the development of nuisance growths of long filamentous algae. In large rivers, water depth of more than 0.4 m provides habitat for brown trout, but in small streams depths in excess of 0.05 m are adequate for most stream insects and native fish (at least for benthic native fish). The flow at which these limiting conditions occur varies with stream morphology. Generally, minimum flow increases with stream size, because stream width increases with stream size. However, the relationship is not linear. In general, small streams require a higher proportion of the natural stream flow to maintain minimum habitat than do large streams.

Minimum flows are not the only influence on fish populations. Studies of trout in the Kakanui River showed that the total adult population was regulated by recruitment and that, in turn, was controlled by the occurrence of floods during spawning and incubation (Jowett 1995, Hayes 1995). The low flows in the Kakanui River had no obvious effect on the trout population; lowest flow in the study period, 0.62 m^3/s, was a little higher than the mean annual low flow of 0.58 m^3/s.

Food availability may limit trout populations, as in the Horokiwi Stream (Allen 1951). Benthic invertebrate biomass was shown to be the single most important factor relating to trout abundance, explaining more than 51% of the variance in 27 different rivers (Jowett 1992a), and in the Kakanui

River the distribution of adult trout mirrored benthic invertebrate abundance, suggesting that it might be a limiting factor (Jowett 1995).

Less is known about the factors controlling native fish populations. New Zealand native fish have evolved to cope with the conditions they experience in our rivers (McDowall 2006). Eels and many galaxiids are able to survive relatively long periods out of water and are capable of some overland movement. Many are also capable climbers and can penetrate to the headwaters of most rivers. The diadromous life history protects their early life stages from the vagaries of the riverine environment. Native fish live at densities of up to about 2 per m^2 in lowland areas, with fish density reducing with elevation. The overwhelming influence of diadromy suggests that total fish numbers and diversity in a given reach will depend on access to the sea, while instream habitat will control the distribution of fish within the reach. Native fish distribution and abundance does not appear to be related to benthic invertebrate abundance (Jowett et al. 1996a), possibly because their food requirement is considerably less than the available food supply and/or because they can survive without feeding for long periods. Therefore, flows that provide adequate native fish habitat are likely to be sufficient to maintain native fish populations (Jowett et al. 2005; Richardson & Jowett 2006).

Juvenile trout, like native fish, occupy shallow water and feed on smaller food items than adult trout. In small tributaries of the Grey River, they were found mostly in shallow water (<0.2 m) with velocities of 0.2–0.6 m/s and their abundance was related to benthic invertebrate biomass (Jowett et al. 1996a).

2.1.1 Stream size and flow requirements of aquatic communities

The composition of the fish community varies with stream size. Small streams are more suited to small fish than large, and vice versa. Small fish have lower swimming speeds and lower velocity and depth preferences than large fish. Adult salmonids usually move upstream or into tributaries to spawn and the juvenile fish rear in these areas, whereas the adults usually move back downstream to deeper waters after spawning. Because water depth and velocity increase with flow, there is usually a flow that provides the best habitat for a particular fish species and life stage. The average habitat suitability index (HSI) at mean annual low flow in 71 New Zealand rivers was calculated for a range of fish species and life stages. When HSI was plotted against flow and a smooth curve fitted for each species and life stage, the peaks of the curves give an indication of the stream sizes that provide the best quality habitat for the species and life stages (Fig. 2.1). Habitat quality increases with flow as streams become wider, until a threshold is reached where further increases in flow result in depths and velocities becoming too high for the species of interest. The optimum size of a river for food producing (benthic invertebrate habitat) was about 15 m^3/s, for adult brown trout 10 m^3/s, and the optimum size for trout fingerlings (< 15 cm) was about 2 m^3/s. This is in agreement with general observations of the distribution of trout, with adult trout in the larger streams and rivers, and trout rearing in small streams or headwaters. The analysis can be extended to native fish and indicates that the optimum size of river for torrentfish, which are common in large braided rivers, is 10–15 m^3/s, whereas streams less than 1 m^3/s contain optimum habitat for many of the other benthic native fish species.

Generalised habitat models, as described in Section 8, also provide a means of determining optimum flow ranges based on habitat suitability and give optimum rivers sizes (e.g., Table 8.1) that are similar to those suggested by this simpler analysis.

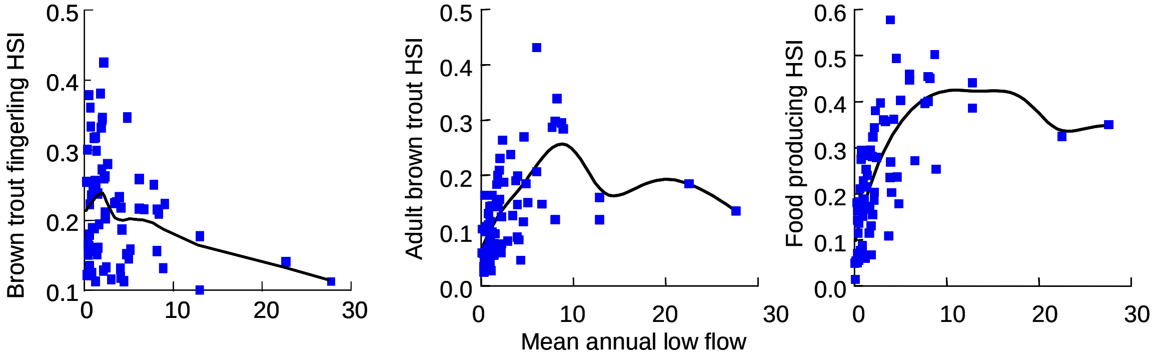

Figure 2.1: **Average habitat suitability index (HSI) at mean annual minimum flow (m^3/s) in 71 New Zealand rivers.**

2.2 Significant elements of the flow regime

Historically, the focus of instream flow studies has been on determining the low flow conditions required to maintain particular instream values, because during low flows there is the greatest competition for the limited amount of water that is available, and the river ecosystem is most under stress. However, several aspects of a river's flow regime may influence its ability to maintain particular instream values. These may be summarised as follows:

- Large floods, in the order of the mean annual flood and greater, are responsible for the overall form of an alluvial river channel. They are known as channel maintenance flows and also influence the nature of the river corridor – the floodplain surface, vegetation cover, and need for river control measures such as willow planting and groynes. Large floods also are a major cause of disturbance to the river ecosystem, with potentially significant impacts – at least for a time – on life-supporting capacity, as aquatic biota are displaced and their habitats temporarily destroyed.

- Smaller floods and freshes, with a frequency of a few times each year, are contained within the channel, and therefore have a more restricted effect than large floods. Nevertheless, they are able to mobilise sediment on at least some areas of the river bed, remove periphyton and other aquatic vegetation, and assist juvenile salmonids and larvae of diadromous native fishes on their passage to the sea. They generally 'flush' and 'refresh' the river bed by removing silt and algal coatings, and inhibit vegetation from colonising the riverbed gravels that are not covered by flowing water. In terms of flow requirements, they are known as flushing flows. As with large floods, the effects of freshes can be both positive and negative – i.e., the effect of flushing and refreshing the river on the one hand, and the effect of disturbing and disrupting parts of the ecosystem on the other.

- Low flows are particularly important because, as suggested above, they are the times at which there is greatest competition for water, the total wetted area of aquatic habitat is least, and the aquatic ecosystem is likely to be under greatest stress (apart from the catastrophic stresses that occur with large floods). On the other hand, stable low flows offer periods of high biological productivity, which permit recolonisation of the riverbed by macroinvertebrates and fish after a flood, and re-establishment of aquatic vegetation.

- Flow variability. The way in which flow varies almost continuously in a river is a significant hydrological feature. Many people consider that flow variations are an essential element of the regime that should be maintained, and that long periods of constant flow ('flat-lining'), which could result from adherence to a minimum flow, should be avoided.

2.2.1 Flow variation

Flow variability is usually considered ecologically desirable, although there has been little scientific measurement of the effect of flow variability on fish populations. In a study of flow variability in New Zealand rivers, Jowett & Duncan (1990) concluded that rivers subject to frequent but relatively small freshes tended to contain fauna that were associated with ‘clean’ rivers, whereas rivers subject to prolonged spells of low flow and less frequent flood flows that were large in comparison with low flows contained invertebrates and algae that were typical of low velocity environments. A river with frequent freshes generally has higher low flows (relative to mean flow) and habitat quality than a river with prolonged low flow and occasional floods

In some countries and for some species, seasonal flow variation is an essential feature for the survival of aquatic life. However, this does not appear to be the situation in New Zealand, where seasonal flow variation is relatively small compared to continental or tropical climates. Similar aquatic communities survive in rivers with very different flow regimes in New Zealand, and this suggests that there is nothing particularly special about the seasonal flow regime of an individual river.

2.3 The relative importance of flow variability versus minimum flow

Before the effect of flow abstraction can be examined, it is necessary to appreciate the inter-relationships between flow variability and the magnitude and duration of low flows. Although flow variability is often thought an essential element of the flow regime that should be maintained, there is little published biological evidence that flow variability is essential. Similar biological communities are often found in streams and rivers with very different patterns of flow variability, and valued biological communities can be maintained in rivers where the flow regime has been extensively modified by hydroelectric operations, such as in the Monowai, Waiau, and Tekapo rivers. The term ‘flow variability’ also confuses the discussion because high flow variability is often bad for the aquatic ecosystem and low flow variability good, depending on how flow variability is defined. Jowett & Duncan (1990) used hydrological indices, particularly the coefficient of variation, to define flow variability. They found that rivers with high flow variability had long periods of low flow and occasional floods, rivers with low flow variability were lake- or spring-fed, and rivers with moderate flow variability had frequent floods and freshes that maintained relatively high flows throughout the year. However, flow variability can also be defined according to the frequency of floods and freshes. Clausen & Biggs (1997) used the frequency of flows greater than three times the median (Fre3) as an index of flow variability and showed, not surprisingly, that periphyton accumulation was less in rivers with more frequent floods (high Fre3), and that invertebrate densities in rivers with moderate values of Fre3 (10–15 floods a year) were higher than those in rivers with high and low Fre3 values. However, as with the Jowett & Duncan (1990) study, the rivers with low Fre3 were also rivers in which there were long periods of low flow without floods.

The effect of flow abstraction on the frequency of floods and freshes and the duration and magnitude of low flows depends on the specific proposals for use of the river – damming, large-scale run-of-river abstraction, or minor abstractions. Potentially, damming can have the greatest effect both on the frequency of floods and freshes and the duration and magnitude of low flows. In fact, damming is the only way the flow regime can be modified sufficiently to affect the channel-forming floods that maintain the character and morphology of the river significantly. Large-scale diversions can increase the duration and decrease the magnitude of low flows significantly and can also reduce the frequency of freshes, but usually have little effect on the channel-forming floods. On the other hand, minor abstractions usually have little effect on the frequency of floods and freshes, even cumulatively, but certainly can reduce flows during periods of low flow.

Flow variability and movement of bed sediments can have profound effects on stream ecosystems. Stable, spring-fed streams are subject to few floods, and the fish and plants that live in such streams are often unable to develop similarly or even to survive in less stable environments. On the other hand, gravel-bed rivers and their aquatic biota are in a constant state of change, caused by extreme flows (floods and droughts) and mobile bed sediments. Floods are an important element of flow variability and flood frequency has been used in several biological models as the primary axis for classifying biological communities (Biggs et al. 1998b). In streams with frequent floods, fish and invertebrates that are small and can colonise new areas rapidly are often dominant (Scarsbrook & Townsend 1993), and the periphyton community is usually sparse, with low species richness and diversity (Clausen & Biggs 1997; Biggs & Smith 2002). In streams with stable flow regimes, aquatic communities are thought to be influenced more by biological processes such as competition between species and grazing/predation than by external environmental factors (Poff & Ward 1989; Biggs et al. 1999).

The biological effects of flow variability usually refer to the effects of floods or the effects of long periods of low flows (e.g., Fig. 2.2). However, we are not aware of any studies that demonstrate that small-scale flow variation is biologically important. In fact, frequent flow variations are usually considered detrimental. Daily and weekly flow fluctuations are often a feature of rivers downstream of hydropower stations. These fluctuations in flow create a varial zone that is wetted and dried as water levels rise and fall. With frequent flow fluctuations, this zone will not sustain immobile plant and invertebrate species. Mobile species such as fish, and probably some invertebrate species, can make some use of this zone, especially for feeding in recently inundated areas of river bed, where there may have been some terrestrial invertebrates in the substrate. However, a varial zone that is wetted and dried at more frequent intervals than a week is unproductive and can be regarded as lost habitat.

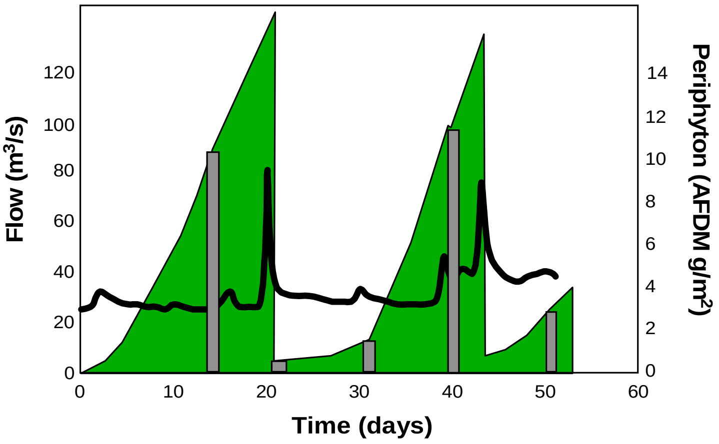

***Figure 2.2*:** **Effect of floods on periphyton accumulation in the Tongariro River (from Jowett & Biggs 1997).**

It can be seen that determining the river flows required to maintain particular instream values may present significant challenges, particularly if there are several values that have different – or even opposite – requirements. Depending on specific proposals for use of the river – damming, large-scale run-of-river abstraction, minor abstractions, etc. – it may be necessary to develop what might be called a ‘designer flow regime’, that considers the need to maintain floods, freshes, low flows, and aspects of flow variability. This, of course, means that the manager must have a clear idea of the outcomes that are desired, with regard to instream values, and the time and resources available to conduct an extensive environmental flow analysis. Although large-scale projects like damming and major diversions will usually require detailed and specific studies to determine downstream flow requirements, minor diversions have little effect on floods and freshes and the main environmental concern will usually be the minimum flow.

3. Instream flow assessment methods

A large number of methods have been used to determine flow requirements and “new” methods continue to be suggested, only a few of which are discussed here. The method or methods used to develop an appropriate minimum flow or flow regime will depend on the case being considered and can vary from a quick rule-of-thumb assessment to detailed studies over several years. Even though methods have been applied for more than thirty years, there is no universally accepted method for all rivers and streams and there are very few cases studies of ecological response to flow changes that can be used to judge the success or failure of different methods.

3.1 Holistic flow assessment

Traditionally, instream flow methods have been used to define a minimum flow, below which no human influences should take place. However, the current trend is away from methods that set one ‘minimum flow’ towards more holistic methods that consider the flow regime and aspects that, with some degree of flow variability, are needed to maintain the natural morphology and ecosystem. Long-term solutions to river flow management need to take a holistic view of the river system, including geology, fluvial morphology, sediment transport, riparian conditions, biological habitat and interactions, and water quality, both in a temporal and spatial sense (Fig. 3.1).

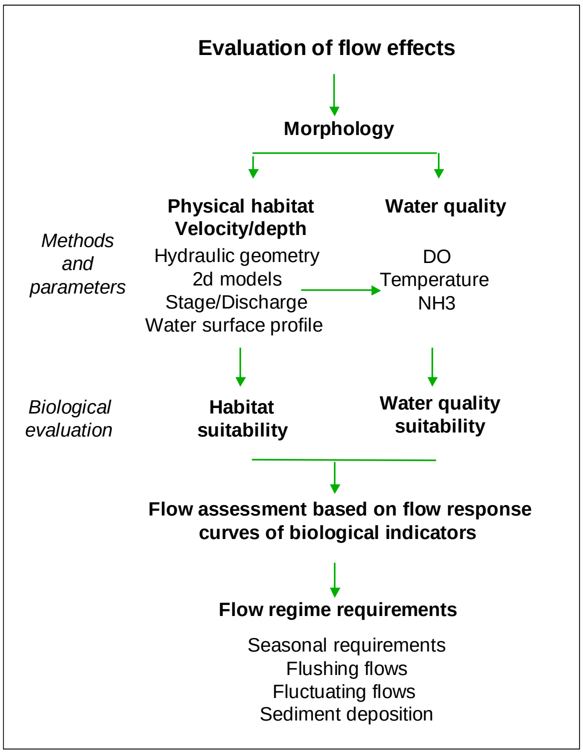

Figure 3.1: **A framework for the consideration of flow requirements.**

The instream flow incremental methodology (IFIM; Bovee 1982) is an example of an interdisciplinary framework that can be used in a holistic way to determine an appropriate flow regime by considering the effects of flow changes on instream values, such as river morphology, physical habitat, water temperature, water quality, and sediment processes (Fig. 3.1). However, a holistic consideration of every aspect of flow and sediment regime, river and riparian morphology, and their associations with the life cycles of the aquatic biota requires a high degree of knowledge about seasonal and life-stage requirements of species and inter-relationships of the various instream values or uses. The holistic framework can be regarded as a checklist against which to evaluate flow changes.

Other flow assessment frameworks are more closely aligned with the 'natural flow paradigm' (Poff *et* al. 1997). The range of variability approach (RVA), and the associated indicators of hydrologic alteration (IHA), allows an appropriate range of variation, usually taken to be one standard deviation, in a set of 32 hydrologic parameters derived from the 'natural' flow record (Richter et al. 1997). The implicit assumption in this method is that the natural flow regime has intrinsic values or important ecological functions that will be maintained by retaining the key elements of the natural flow regime. Arthington et al. (1992) described an 'holistic method' that considers not only the magnitude of low flows, but also the timing, duration and frequency of high flows. This concept was extended to the building block methodology (BBM), which 'is essentially a prescriptive approach, designed to construct a flow regime for maintaining a river in a predetermined condition' (King et al. 2000). It is based on the concept that some flows within the complete hydrological regime are more important than others for the maintenance of the river ecosystem, and that these flows can be identified, and described in terms of their magnitude, duration, timing, and frequency.

In concept, the BBM is similar to the IFIM in aiming to maintain a prescribed condition based on a high degree of knowledge about flow requirements of the various aspects of the ecosystem. However, identification of flow requirements in the BBM is based more on the 'natural flow paradigm' than on an understanding of physical and biological relationships. A basic assumption of the BBM, and the major point of departure from IFIM, is that biota associated with a river can cope with naturally occurring low flows, that occur often, and may be reliant on higher flow conditions. Furthermore, flows that are not characteristic of the river will constitute an atypical disturbance to the ecosystem and could fundamentally change its character (King et al. 2000).

3.2 Historic flow methods

These methods are based on flow records and are the simplest and easiest to apply. Stalnaker et al. (1995) describe this type of method as 'standard setting' because they are generally desktop rule-of-thumb methods that are used to set minimum flows. A historic flow method is based on the flow record and uses a statistic to specify a minimum flow, below which water cannot be abstracted. The statistic could be the average flow, a percentile from the flow duration curve, or an annual minimum with a given exceedance probability. For example, a method might prescribe that the flow should never drop to 30% of the mean annual low flow (MALF), or it could recommend that the average flow should stay above 80% of MALF. The percentage used is referred to as the 'level of maintenance'.

The aim of historic flow methods is to maintain the flow within the historical flow range, or to avoid the flow regime from deviating largely from the natural flow regime. The underlying assumption is that the ecosystem has adjusted to the flow regime and that a reduction in flow will cause reduction in the biological state (abundance, diversity, etc.) proportional to the reduction in flow; or in other words, that the biological response is proportional to flow (Fig. 3.2). It is usually also assumed that the natural ecosystem will only be slightly affected as long as the changes in flow are limited and the stream maintains its natural character. It is implicitly assumed that the ecological state cannot improve by changing the natural flow regime.

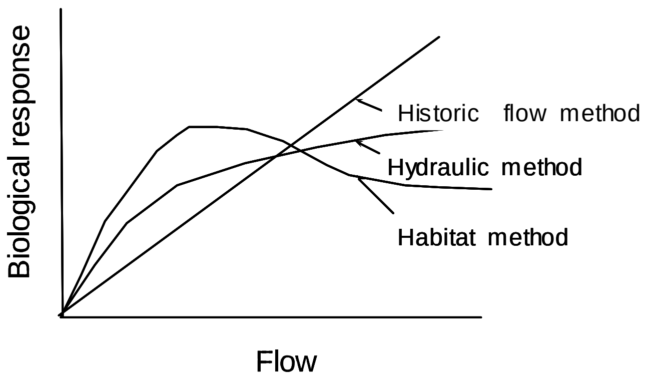

Figure 3.2: **Hypothetical relationships between assumed biological response to flow for the historic flow, hydraulic and habitat methods. The biological response is assumed to be proportional to the flow, the wetted perimeter or width, and the weighted usable area, for the historic flow method, the hydraulic method, and the habitat method, respectively.**

The most well known historic flow method is the Tennant (1976) method, also known as the Montana method, which specifies that 10% of the average flow is the lower limit for aquatic life, and 30% of the average flow provides a satisfactory stream environment. The Tennant method was based on hydraulic data from eleven U.S. streams (Montana, Wyoming, and Nebraska) and an assessment of the depths and velocities needed for sustaining the aquatic life. At 10% of average flow, he found that the average depth was 0.3 m and velocity 0.25 m/s, and considered these lower limits for aquatic life. He found that 30% of average flow or higher provided average depths of 0.45–0.6 m and velocities of 0.45–0.6 m/s and considered these to be in the good to optimum range for aquatic organisms. This is an example of a ‘regional method’, applicable to the region that has the same type of streams as the streams used for developing the method. However, the Tennant method has been adopted in many different parts of the world, including New Zealand, and in some cases, its recommended minimum flows have been similar to IFIM predictions (e.g. Allan 1995; Crowe et al. 2004). In New Zealand, Fraser (1978) suggested that the Tennant method could be extended to incorporate seasonal variation by specifying monthly minimum flows as a percentage of monthly mean flows.

Historical flows can also be used to define ‘an ecologically acceptable flow regime’; for example, Arthington et al.'s (1992) ‘holistic method’ that considers the magnitude of low flows, and the timing, duration and frequency of high flows. Such a flow regime would not only sustain biota during extreme droughts, but would also provide high flows and flow variability needed to maintain the diversity of the ecosystem. The building block method (BBM; King et al. 2000) is a similar approach. The holistic, BBM and RVA methods are conservative and maintain the ecosystem by retaining the key elements of the natural flow regime, but have not been widely used (Tharme 2003). These are ‘low risk’ approaches aimed at maintaining an ecosystem in its existing state and preclude the possibility that a river ecosystem can be enhanced by other than a natural flow regime. They are probably most appropriate for river systems where the linkages between ecosystem integrity and flow requirements are poorly understood.

3.3 Hydraulic geometry methods

Channel shape is determined primarily by geology and the flow regime of a river. The relationship between hydraulic geometry and flow can be defined between rivers or sites on rivers, using downstream or at-a-site hydraulic geometry, respectively; the latter is also known as at-a-station geometry. For alluvial rivers, downstream hydraulic geometry relationships between channel form and flow are similar in rivers worldwide (e.g., Leopold & Maddock 1953; Kellerhals & Church 1989). River width increases with the square root of discharge (exponents range from 0.45–0.54; Park 1977;

Kellerhals & Church 1989; Jowett 1998). Water depth and velocity also increase with discharge, although the relationships are not as well defined. At a site hydraulic geometry relationships are more variable and less well reported. For New Zealand rivers, Jowett (1998) gives the average relationships at a site as:

$$W \propto Q^{0.207}$$

$$D \propto Q^{0.335}$$

$$V \propto Q^{0.458}$$

where Q is the discharge, W the average width, D the average water depth, and V the average velocity. These at a site relationships are averages derived over low to normal flow ranges. For any particular river, the exponent of the relationship can change if there is an abrupt change in geometry, such as at the point where a river overflows its banks onto its floodplain. These abrupt changes in geometry will correspond to breakpoints (sometimes incorrectly called inflection points) of width/flow or depth/flow curves (e.g. Mosley 1992). Breakpoints in the relationships between width, depth, or habitat and flow are usually well defined in rivers of moderate gradient in well-defined channels. Braided rivers are more problematical. As flows increase, additional braids form, increasing width and usable habitat, until the wide gravel flood plain is inundated (Mosley 1982). In this situation there are no clear breakpoints, at least not in the low to median flow range.

When hydraulic geometry is used as a flow assessment method, the analysis is usually based on measurements of hydraulic data (wetted perimeter, width, depth or velocity) from one or several cross-sections in the stream. The aim of hydraulic methods is to maximise food production by keeping as much as possible of the food-producing area below water. Because the streambed is considered the most important area for food production (periphyton and invertebrates), it is usually the wetted perimeter or the width that is used as the hydraulic parameter.

The variation of the hydraulic parameter with flow can be found by carrying out measurements at different flows, or from calculations based on rating curves or Manning’s equation. The graph of the hydraulic parameter versus flow (Fig. 3.2) is used for prescribing recommended flows, or to specify a minimum flow. The minimum flow can be defined as the flow where the hydraulic parameter has dropped to a certain percentage of its value at mean flow, or the flow at which the hydraulic parameter starts to decline sharply towards zero (the curve’s breakpoint). If the wetted perimeter or width is used, the breakpoint is usually the point at which the water covers just the channel base. However, wetting of the channel base might not be enough to fulfil the requirements to depth and velocity for some species.

Gippel & Stewardson (1998) suggest an objective method for defining a breakpoint in wetted perimeter/flow (P/Q) relationships, that could be very useful for maintaining consistency in flow assessments between rivers. They suggested the breakpoint could be selected as either the point of maximum curvature or the point where the slope (dP/dQ) is 1, after first normalising wetted perimeter and flow by dividing by their respective values at an index flow, such as the median flow.

3.4 Habitat methods

Of the three basic types of instream flow methods, historic flow methods are coarse and largely arbitrary, unless the natural flow paradigm is adopted and historical flows are specified so that they mimic natural flows. Hydraulic geometry methods provide information on the physical characteristics of the river, but do not have strong links to biological requirements. Habitat methods are an extension of the hydraulic methods. Their great strength is that they quantify the loss of habitat caused by changes in the natural flow regime, which helps the evaluation of alternative flow proposals.

The aim of habitat-based methods is to maintain, or even improve, the physical habitat for instream values, or to avoid limitations of physical habitat. They require detailed hydraulic data, as well as knowledge of the ecosystem and the physical requirements of stream biota. The basic premise of habitat methods is that if there is no suitable physical habitat for the given species, then they cannot exist. However, if there is physical habitat available for a given species, then that species may or may not be present in a survey reach, depending on other factors not directly related to flow, or to flow related factors that have operated in the past (e.g., floods). In other words, habitat methods can be used to set the 'outer envelope' of suitable living conditions for the target biota.

Biological information is supplied in terms of habitat suitability curves for a particular species and life stage. A suitability value is a quantification of how well suited a given depth, velocity or substrate is for the particular species and life stage. Other relevant factors, such as cover, aquatic vegetation and presence of other species, can be incorporated into the evaluation of habitat suitability, although this is not common.

The result of an instream habitat analysis is strongly influenced by the habitat criteria that are used. If these criteria specify deep water and high velocity requirements, maximum habitat will be provided by a relatively high flow. Conversely, if the habitat requirements specify shallow water and low velocities, maximum habitat will be provided by a relatively low flow and habitat will decrease as the flow increases. In contrast to historic flow methods, the habitat method does not automatically assume that the natural flow regime is optimal for all aquatic species in a river.

Habitat methods and water quality models can be integrated, although usually the results of hydraulic models are transferred into water quality models. For example, a water temperature model (SSTemp; Bartholow, 1989) uses measurements or estimates of water depth and velocity for each flow and these data are then used to model how water temperature varies with distance downstream. The integration of stream geometry and water temperature, dissolved oxygen and ammonia models has been implemented in the decision support system WAIORA (Jowett et al. 2003).

The two key elements of a habitat based method are the habitat suitability criteria that are used to calculate habitat, and the linkage between available habitat and aquatic populations. These two issues can be discussed and argued without resolution, although the bottom line is that there must always be suitable habitat if an aquatic species or use is to be maintained. An ecological justification can be argued for the MALF (see Section 2 Stream Ecology), and the concept of a low flow habitat bottleneck for large brown trout has been partly justified by research (e.g. Jowett 1992a), but setting flows at lower levels, such as the 7-day 5 year low flow ($Q_{7,5}$) or $Q_{7,10}$ is rather arbitrary. Hydraulic methods do not have a direct link with instream habitat and interpretation of ecological thresholds based on breakpoints or other characteristics of hydraulic parameters, such as wetted perimeter and mean velocity, are arbitrary and depend on rules of thumb and expert experience. On the other hand, habitat based methods have a direct link to habitat use by aquatic species. They predict how habitat (as defined in by various habitat suitability models) varies with flow and the shapes of these characteristic curves provide the information that is used to assess flow requirements. Habitat based methods allow more flexibility than historic flow methods offering the possibility of allocating more flow to out-of-stream uses while still maintaining instream habitat at levels acceptable to other stakeholders (i.e., the method provides the necessary information for instream flow analysis and negotiation).

3.5 Regional methods

Tennant's (1976) method is a good example of a regional method that combines the best features of historic flow methods and habitat methods, resulting in a biologically defensible method of minimum flow assessment – for the region. Once established, regional methods can be easily applied to rivers within the region using a formula based on the proportion of natural flow, either recorded or estimated. The formula can be as simple as a fixed proportion of flow or can vary the proportion with river size, possibly retaining a higher proportion of the flow in small rivers than in larger rivers, as used in formulae for maintenance of trout and food producing habitat in Wellington and Taranaki rivers (Jowett 1993a,b). Similar methods could be developed for regions that are hydrologically and morphologically similar, with criteria that apply to trout, native fish, stream insects, or periphyton. By analysing habitat variation with flow for rivers within a region, it is possible to determine the level of flow as a proportion of median or mean annual low flow that maintains adequate or optimum conditions for various 'target' communities. Variation in levels of maintenance could be achieved by assessing requirements for optimum habitat and minimum habitat, as in the Tennant method. Application of the method would involve selecting an appropriate target community and level of maintenance for the river in question and then applying a formula based on flow.

The benefit of regional methods over historic flow methods is that they can have explicit environmental goals, making water management more transparent. Thus, regional methods can be established as biologically defensible, and discussion and consultation can focus on whether the 'target' and flow standards of maintenance are appropriate.

The rationale for habitat based regional methods is primarily that of habitat methods. Within a region, it is possible to develop formula that predict when hydraulic conditions are optimum or become limiting for a range of aquatic species. For instance, most native fish are small stream species. Few are found in swift, deep water. In contrast, adult trout are rarely found in water less than about 0.4 m deep. Stream insects are most abundant in shallow swift habitats.

It is also possible to generalise velocity and depth criteria as levels of protection within a region, based on a data set from rivers in the region. For instance, average velocities of less than 0.1 m/s might be considered poor, 0.1–0.3 m/s adequate, and 0.3–0.5 m/s good for aquatic organisms such as trout and benthic invertebrates. Similarly, average depths greater than 0.15 m might be considered suitable for native fish and depths greater than 0.4 m suitable for adult trout.

These methods are potentially useful in that they combine the best features of habitat and flow methods. Once developed, they are less expensive than habitat methods, yet are still likely to result in flow assessments that provide life sustaining flows, whilst retaining some degree of the river's 'character'.

3.6 WAIORA

WAIORA, Water Allocation Impacts on River Attributes, (Jowett et al. 2003) is a decision support system that uses information on stream morphology, either from simple measurements at two flows or from a RHYHABSIM dataset, to predict how instream habitat, dissolved oxygen, total ammonia, and water temperature change with flow. Although WAIORA does not incorporate habitat suitability curves, the generalised models described in Section 8 can be easily implemented, either in the programme or as an additional calculation. WAIORA calculates the effects of flow on instream habitat, dissolved oxygen, total ammonia, and water temperature, and links the output to environmental guidelines that can be specified by the user to determine if an adverse effect is likely to occur. A number of assumptions have been made during model development and these are detailed in a manual and help file. The outputs of WAIORA reflect the nature of these assumptions and the quality of the data entered by the user. The models are better at predicting the relative amount of change associated with flow scenarios than at predicting absolute changes. Some guidance on the expected accuracy of models and 'comfort zones' associated with guideline thresholds is provided in the help file and the summary plots produced by the WAIORA program.

4. Instream values and management objectives

4.1 Defining instream values

Instream values may be grouped into:

- ecological or intrinsic values
- landscape, scenic and natural characteristics of the river
- angling and fishing values
- amenity values – boating and other recreational activities undertaken in, on or near the river
- Māori values.

There are, of course, overlaps and linkages among these values. They differ in the extent to which they are influenced by variations in flow regime. ‘Flow-related values’ change in a discernible way as flow changes. For example, the value of a particular river as a fishery may decline as flow declines, because the area of suitable habitat declines. At the other end of the scale, increasing flow also may make the river increasingly unattractive for angling, and there can be a range of flows that is preferred or optimal for the sport (Carlson & Palmer 1997; Hayes & Young 2001). ‘Flow-independent values’ change to a minor extent, or not at all, as the flow changes. Factors like water quality, water temperature and the micro-distribution of turbulence and velocity change with flow, but the flow-related changes are often small and the biological effects are difficult to predict because of the large natural variation in these factors and the wide tolerances of aquatic organisms.

Sustaining instream values when there is demand for out-of-stream water use is challenging for water resource managers. ‘Sustain’ means different things to different people. Moreover, it is difficult to sustain all values at original levels when flows change. It is naïve to expect that instream habitat conditions and the stream ecosystem will remain exactly the same once a flow regime is altered. It also needs to be appreciated that there often is no clearly identifiable point at which instream conditions become untenable as flows are reduced, except when rivers cease flowing. In the face of this knowledge, the challenge is to determine the degree of change in flow and instream conditions that can occur before instream values are eroded noticeably and reach levels that dissatisfy community interests. Science, presently, can provide only partial answers for this problem. As a result, some of the decision making is necessarily arbitrary and influenced by stakeholder politics.

4.2 Management objectives

A basic principle established in the *Flow Guidelines* (Ministry for the Environment 1998) is that instream values and their requirements must be identified and appraised within the context of definite instream management objectives (Fig. 4.1). Without these, instream values that are expressed in (non-monetary) environmental or amenity terms may receive less consideration than out-of-stream uses of water, whose values can be expressed in terms of dollars. However, where objectives have been developed consultatively to reflect community aspirations, they can be accorded appropriate weight, even though they might not be expressed in monetary terms. Resource management objectives have been defined by regional councils in their various regional policy statements, and an increasing number of councils are developing more specific objectives in regional or catchment water resource management plans. These objectives provide a reference point from which council officials, special tribunals, or the Environment Court can compare the merits of alternative uses of a given body of water, and in particular the extent to which instream values must be provided for.

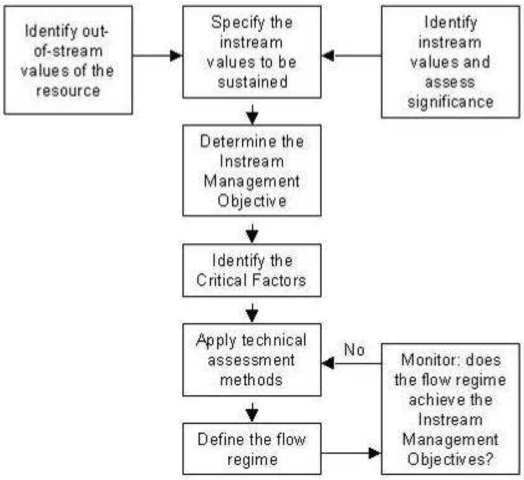

Figure 4.1: **The process of setting objectives for management of instream flow regimes (from Ministry for the Environment 1998).**

Instream flow management is a complex process, usually involving a combination of technical, public, and legal considerations. To be effective, the instream flow management process should consider the present status of the river and its ecosystem, and then, in consultation with public and institutional organisations, set goals and objectives before establishing appropriate flow requirements. Instream flow methods play a part in this process by showing how the requirements of instream uses (in terms of their various parameters, such as wetted perimeter, instream habitat, and water quality) vary with flow. Once these relationships are established, the next important decision is the level at which instream values should be maintained. This is relatively simple where there are established water quality standards, such as for dissolved oxygen and ammonia. However, acceptable levels of instream habitat and even water temperature are more difficult to decide. The *Flow Guidelines* (Ministry for the Environment 1998) suggest that the level of maintenance should reflect the merits of instream values in a particular river (e.g., the quality of a recreational fishery, the biological diversity of a stream ecosystem, the conservation status of a breeding bird population on a river bed, the proximity to a large population centre of a kayaking river, the availability of alternatives or means of mitigation, etc). The concept of retaining a percentage of the ‘natural’ condition is one means of defining the level of maintenance, with the proportion of habitat retained varying according to the merits of the instream values and community aspirations.

The management process needs to specify:

1. resource definition and assessment of instream values
2. clear goals and target objectives, and
3. defined levels of protection.

Failure to do this will lead to conflicting minimum flow requirements. Attempts to maintain everything in the existing state invariably lead to the conclusion that flows should not be changed and precludes the opportunity for enhancement of some aspects of the aquatic environment, and use of the water resource.

4.2.1 Critical values

The concept of critical values is that by providing sufficient flow to sustain the most flow sensitive, important value (species, life stage, or recreational activity), the other significant values will be also be sustained. ‘Because it is unrealistic to expect that all values will be maintained at original levels when flows change, ‘sustain’ should be taken as meaning maintaining critical instream values at levels not noticeably different to existing levels and to the satisfaction of stakeholders. Identification of the critical instream values and appropriate standards of maintenance are an essential basis for the assessment of instream flow requirements. The critical values must be appropriate to the stream, particularly its size, and must be related to flow, particularly minimum flows, if habitat-based methods are to produce consistent and sensible results.

The critical values and their associated habitat suitability criteria can be perceived in two ways. In most cases, we apply them in a specific sense for providing habitat for the target critical species/life stage and with the added aim of providing for taxa with lower flow requirements. In some combinations of stream source and flow range, we use the habitat criteria associated with the critical value in a generic sense of providing general instream conditions that, based on experience, we consider appropriate for the ecological function and potential range of instream communities. In this latter situation, the habitat criteria act as general descriptors of instream conditions and stream size; the ‘target species’ is secondary and may in fact not actually be present. Examples of these applications include:

- trout spawning criteria which also provide good depths and velocities for invertebrate habitat (which sustains the fish food base) in small streams
- redfin and common bully habitat criteria that provide good general instream conditions for streams slightly larger than those dominated by diadromous galaxiids.

Critical values can also be selected according to the fish communities present in a river (see Richardson & Jowett 2005 for more detail on communities and their stream habitats), for example:

- trout spawning and juvenile rearing
- large adult trout
- non-migratory galaxiids.
- diadromous galaxiids (inanga, giant and banded kokopu)
- redfin bully/common bully gravel bed community.

In slow-flowing environments, water quality may be a significant determinant of stream ecology. The effects of flow on water quality can be estimated using the decision support system WAIORA (Jowett et al. 2003).

4.2.2 Critical values as substitutes for other significant values

In New Zealand, it has generally been assumed that minimum flows set for salmonids will be adequate to maintain native fish populations. The rationale for this is that trout, because of their large size and drift-feeding requirements, have higher depth and velocity requirements than most native fishes. Many native fishes are most abundant in small streams or on the margins of larger rivers (e.g., upland bullies, redfin bullies, inanga). Therefore, habitat for these species is maximal at low flow. The river margins will still provide some habitat for these native fishes at the higher flows required by salmonids.

The fast water habitat native fish guild comprising torrentfish and bluegill bully have similar flow requirements to adult trout. Optimum habitat for these species, especially for torrentfish, typically occurs at high flows (Fig 4.2). Similarly, optimum flows for some native invertebrate taxa occur at higher flows than trout (Fig. 4.2). Nevertheless, flow conditions usually are not set for these fast water species alone because they do not have sufficiently high value. Torrentfish and bluegill bullies are relatively common and widespread and do not support fisheries (McDowall 2000). Furthermore, it is inappropriate to set minimum flows to maximise torrentfish and aquatic invertebrate habitat because such flows can not be sustained by the natural flow regime in smaller rivers or in rivers where the flow spreads out over a wide gravel flood plain. Flows that sustain maximum habitat for these fish usually are higher than the natural mean annual low flow and may be higher than the median flow.

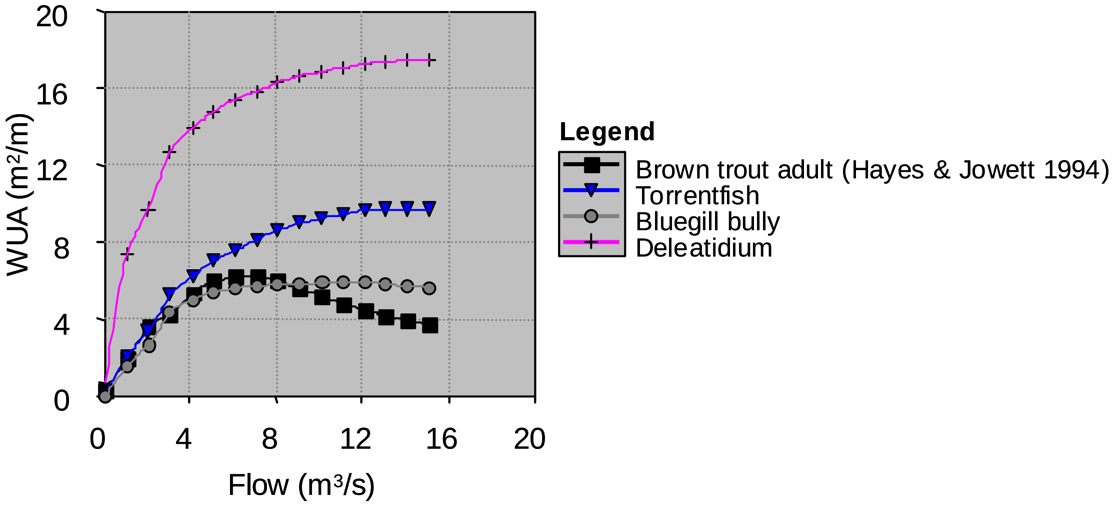

***Figure 4.2*:** **Relationships between instream habitat (WUA) and flow for fast water guild native fish (torrentfish and bluegill bully), adult brown trout and the mayfly *Deleatidium* for the Oreti River at Centre Bush.**

The other native fish and invertebrate species are widespread and relatively common in most rivers, and many of the fish species do not have fisheries values. The relevant flow management aim for these species is maintenance of biotic natural character, perhaps using the native fish species as an indicator of biotic value. Therefore, it may not be necessary to provide flows that sustain maximum habitat or potential maximum abundance. Moreover, many of the native fishes have life history features that impart resilience to environmental change. A large percentage of the native fish fauna in a given river reach is likely to be diadromous, especially close to the sea. These populations probably are recruited from a common gene pool – at least at the regional level. Therefore, environmental change in a given river may not necessarily affect recruitment of the population. Some of the common resident native fish species have a high intrinsic rate of population increase, a feature that is well suited to variable flow conditions (e.g., upland bully).

Nevertheless, there are situations in which the conservation status of certain native fish species warrants special attention. These concern some of the non-migratory galaxiids and large diadromous galaxiids (giant, shortjaw, and banded kokopu). Usually these species do not co-occur with trout. These galaxiids all have lower flow requirements than trout; and in addition to flow, they may require other features, including riparian and instream cover, and preferably native forest in the catchment or on the stream margins.

Maintenance of trout habitat ought to favour shags by maintaining trout populations that shags can exploit for food. The feeding habitat requirements of wading birds, terns and gulls should be adequately provided for by the maintenance of adult and juvenile trout habitat – inasmuch as the habitat of aquatic invertebrates is taken into consideration in the maintenance of the latter. Birds and trout rely on aquatic invertebrate production in shallow riffles and runs for food. If the concept of sustaining productivity of trout populations is an integral part of a minimum flow regime then the food requirements of birds ought to be well catered for.

A recent study by Tipa & Teirney (2003) identified Māori values for streams in the Otago region. It showed that some of the values identified by Māori were highly correlated with biological measures of stream health, such as the macro-invertebrate community index (MCI) and a similar index described by Biggs et al. (1998a). This relationship with biological indices of stream health suggests that flow recommendations that maintain healthy invertebrate communities would maintain Māori values, at least partly. However, Tipa & Teirney (2003) and the Ministry for the Environment's Flow Guidelines (1998) suggest iwi participation in the determination of a suitable flow regime.

4.2.3 Levels of maintenance

Levels of habitat maintenance provided by minimum flows are usually set arbitrarily. This is partly because our state of knowledge on the effects of low flow is insufficient to predict the response of stream ecosystems, and particularly fisheries, and partly because instream habitat simply declines continuously as flow falls below the optimum value, at least in streams and smaller rivers. Therefore, there is no clearly identifiable point at which instream conditions become good or bad, but rather habitat simply gets worse as flow falls below the optimal value – although the rate of habitat change may vary with flow. When habitat modelling results are available, the rate of change of habitat is often used as a basis for setting a minimum flow. The point of greatest change in the rate (the breakpoint) is often selected as the minimum flow. This is based on the premise that higher flows offer diminishing benefits for instream habitat, although there is no scientific evidence that the breakpoint is correlated with biological response. In assessing the amount of habitat to be retained at low flow, it is important to realise that if the low flow were to provide maximum habitat, then higher flows would provide less than maximum habitat. Such a situation may be less than optimum for the species in question, although the risk of detrimental effect of increasing the flow above that which provides maximum habitat is not as great as decreasing the flow, and any habitat loss may be balanced by an increase in food production or the amount of cover. The 'best' brown trout rivers, such as the Mataura and Motueka, have flows that provide near maximum habitat between the mean annual low flow and the median flow.

Instream habitat modelling can estimate the incremental (or percentage) reduction in habitat as flow declines. This can assist stakeholder negotiation over minimum flows where it is useful to consider the relative values of instream versus out-of-stream values in the negotiation. However, how much habitat reduction is enough is more a matter of arbitrary stakeholder choice rather than ecological science.

Levels of habitat retention are conservative, in that we believe that they are unlikely to be proportional to a population response. Theoretically, a change in available habitat will only result in a population change when all available habitat is in use (Orth 1987). In most cases, population densities are probably at less than maximum levels because flows are varying all the time. That being the case, a habitat retention level of, say 90%, would maintain existing population levels, whereas retention levels of 50% might result in some effect on populations, especially where densities were high.

It is also possible to vary the level of habitat retention according to the significance of instream and out-of-stream values. An arbitrary habitat reduction from the maximum value might provide the basis for the minimum flow decision, taking into account the relative importance of instream versus out-of-stream values. As with critical values, the categories and levels by which habitat retention levels could be adjusted for each of the categories should be set in consultation with the community and stakeholders.

Critical values and out-of-stream uses will need to be assessed on a catchment basis, because the significance of critical values may change as the river flow increases. Small tributaries may have low significance ratings yet contribute to the flow of a river with high ratings downstream. Maintenance of a minimum flow at the downstream site may depend on adequate flows in smaller tributaries. The flow assessment procedure should evaluate flow requirements at points along the stream network to identify the most downstream location with the highest flow demands. Ideally, this would be used as a monitoring site so that when flows at this site reach a minimum, water restrictions would be applied to all upstream consents.

5. Hydraulic habitat modelling

Overall objective of hydraulic habitat modelling

To create hydraulic conditions that will sustain the ecosystem in a prescribed condition in terms of:

- hydraulic habitat
- water quality
- flow regime.

5.1 Instream flow incremental methodology

The combination of a description of habitat suitability with hydraulic modelling of river flow is hydraulic habitat modelling, and is the main component of the instream flow incremental methodology or IFIM (Bovee 1982). Hydraulic habitat modelling is also known as instream habitat modelling or physical habitat modelling. The models are of physical habitat (water depth and velocity) and apply instream, so the term hydraulic encompasses both. Although the best known physical habitat model (PHABSIM) was limited to prediction of physical habitat (depth, velocity, and substrate), hydraulic habitat models can also predict the effect of flow on water temperature and dissolved oxygen concentration. They provide a means of condensing diverse data into a result that describes how the amount of instream habitat changes with flow.

5.2 Habitat and hydraulic spatial scales

Habitat can be defined at different spatial scales. It is used to describe the location and environmental conditions where organisms live, or where they could live (usually termed microhabitat). However, it is also used to describe a general area, such as riffle habitat (mesohabitat) or even broader conditions, as in aquatic and terrestrial habitats (macrohabitat). Physical or hydraulic habitat describes the physical instream conditions (usually water depth, velocity and substrate) and does not consider biotic or water quality conditions. Here, suitable or preferred habitat is used to describe the range of physical conditions in which and organism is most likely to be found.

The aim of a minimum flow is to retain adequate water depths and velocities in the stream or river for the maintenance of aquatic life and other instream uses. Instream habitat models predict the flows necessary to maintain, or even improve, the physical habitat for target biota, or to avoid limitations of physical habitat. Because the purpose of hydraulic models is to predict physical habitat, the scale at which habitat is defined by the habitat suitability criteria and the scale of hydraulic model predictions should be similar. For example, if velocities for the derivation of habitat suitability criteria were measured at 0.6 of the depth, the hydraulic model should predict velocities at the same depth.

There is some confusion about the scale at which hydraulic habitat models work. Although they are often claimed to predict microhabitat, they do not truly predict the range of velocities experienced in a river. For example, they do not predict the eddies and currents that surround a boulder. However, such currents and eddies depend on depth of water and average column velocity and suitable microhabitats will be provided by the larger scale hydraulic conditions. Thus, these models essentially consider habitat at a meso- to macrohabitat level rather than microhabitat level, maintaining suitable depths and average velocities, and a degree of habitat diversity that is generated by the morphology of the river and is largely independent of flow.

5.3 Hydraulic habitat modelling process

The first hydraulic habitat methods (e.g. McKinley 1957) used simple hydraulic modelling, or surveys at different flows, to determine the flows that provided maximum salmonid spawning areas – areas with gravel substrate, with water depths of 0.2–0.4 m and velocities of 0.2–0.7 m/s (Smith 1973). After this, the methods began to get more complicated, with multiple options for hydraulic modelling and habitat evaluation (Milhous et al. 1989). Of the available methods for minimum flow assessment, habitat based methods are the most justifiable because of their simple yet defensible basis of providing suitable habitat for aquatic species.

Hydraulic habitat models are used to predict habitat changes with flow, and to assist decisions on an acceptable flow regime, usually with an emphasis on minimum flow requirements. These models predict water depth, velocity, and other hydraulic variables for a range of flows and then evaluate habitat suitability. Current hydraulic habitat models include PHABSIM (physical habitat simulation; Bovee, 1982; Milhous et al. 1989), RHABSIM (river habitat simulation), RHYHABSIM (river hydraulic habitat simulation; Clausen et al. 2004), EVHA (evaluation of habitat; Ginot 1998), CASIMIR (Jorde 1997), RSS (river simulation system; Killingtviet & Harby 1994), River2D (2D model; Ghanem et al. 1996; Waddle et al. 2000; Steffler et al. 2003), Hydro2dE (2D model; Beffa 1996; Duncan & Carter 1997)), SSIIM (3D model; Olsen & Stokseth 1995).

The use of these models requires detailed hydraulic data, as well as knowledge of the ecosystem and the physical requirements of stream biota. The basic premise in evaluation of flow requirements is that if there is no suitable physical habitat for the given species, then they cannot exist. However, if there is physical habitat available for a given species, then that species may or may not be present in a survey reach, depending on other factors not directly related to flow, or to flow related factors that have operated in the past (e.g., floods). In other words, habitat can be used to set the ‘outer envelope’ of suitable living conditions for the target biota.

Hydraulic habitat models can be separated into a hydraulic component and a habitat component. The hydraulic model predicts water velocity, depth and other hydraulic variables at a given flow, for each point, represented as a cell in a grid covering the stream area under consideration. In addition, information on bed substrate and other relevant factors such as shade, aquatic vegetation and temperature, can be recorded for each cell.

Biological information for the habitat component is supplied in terms of habitat suitability criteria (or curves) for a particular species and life stage. A suitability value is a quantification of how well suited a given depth, velocity or substrate is for a particular species, size, life stage, and behaviour.

The result of an instream habitat analysis is strongly influenced by the habitat criteria that are used. Selection of appropriate criteria and determination of habitat requirements for an appropriate flow regime requires a good understanding of the species’ life cycles and food requirements (Heggenes 1988; 1996).

The hydraulic habitat analysis starts by choosing a particular species, size, life stage and behaviour and defining suitability criteria. Waters (1976) proposed the use of a suitability index that varies between 0 (unsuitable) and 1 (optimal) as an alternative to binary criteria (0 unsuitable or 1 suitable) that had been used by in earlier hydraulic habitat studies (McKinley 1957; Collings 1972). Intuitively, it seems reasonable to consider conditions that are of intermediate habitat value, between optimal and barely useful. For each point in the survey (Fig. 5.1), velocity, depth, substrate, and possibly other parameters (e.g., cover) at the given flow are converted into suitability indices, one for each parameter. The suitability indices can then be combined (usually they are multiplied) and multiplied by the area that they represent to give an area of usable habitat. Finally, all the usable habitat areas can be summed to give the weighted usable area (WUA m^2/m) for the reach at the given flow. If the suitability is >0, the point will contribute to the total area, but if it is zero the point makes no contribution. This whole procedure is then repeated for other flows to produce a graph of WUA versus flow for the given species. This graph has a typical shape, shown in Figure 5.2 with a rising part, a maximum and then may decline. The decline occurs when the velocity and/or depth exceed those preferred by the given species and life stage. Thus, in large rivers, the curve may predict that physical habitat will be at a maximum at flows less than occur naturally.

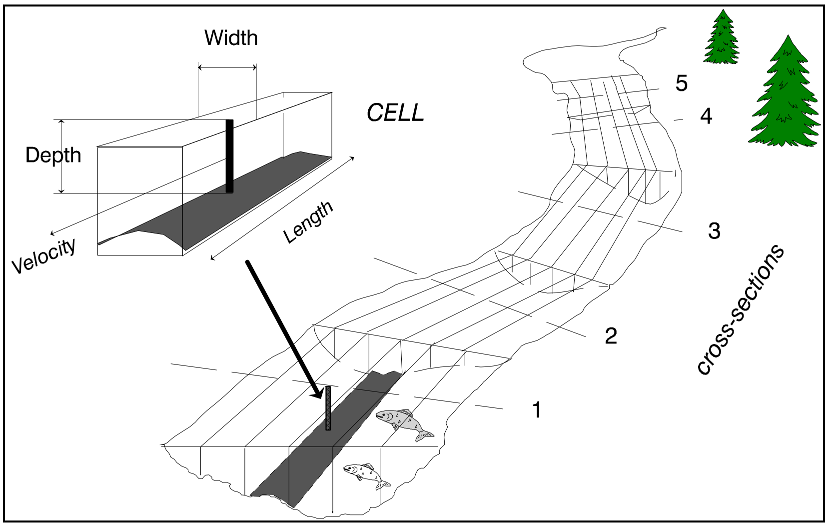

***Figure 5.1*: Representative reach habitat survey of a stream reach, showing the area represented by a point measurement.**

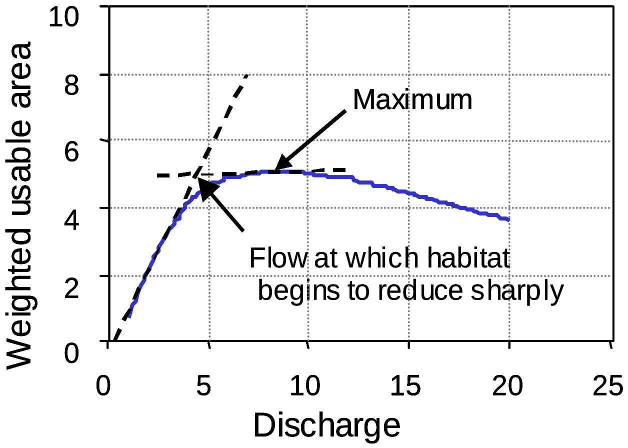

***Figure 5.2*: Selection of minimum flow at the point where habitat begins to decline sharply with decreasing flow.**

The fundamental criticism of IFIM (PHABSIM) by Mathur et al. (1985) and Scott & Shirvell (1987) was that there was no evidence that there was any correlation between species abundance and the amount of suitable habitat. Since then, some studies have demonstrated relationships between WUA and species abundance and in some cases, such as benthic invertebrates, suitability is derived by fitting a curve to the relationship between the habitat variables and species abundance, so that there is an inherent correlation between predicted habitat suitability and species abundance, as shown in Jowett (1992b) and Jowett & Davey (2007). In cases where habitat suitability curves are based on presence/absence data, WUA will be an index of the probability of use and will predict the relative distribution of fish in a reach, as shown by Hardy et al. (1983). However, the warning is valid and use of inappropriate habitat suitability curves could give misleading results. It is also necessary to consider all requirements for a species' continued survival. For example, the primary requirements for salmonids are both space and food (Chapman 1966), so assessment of instream flow needs for salmonids must consider both space (i.e., habitat) and food requirements. The relationship between habitat and flow (Fig.5.2) can be used to define a preferred flow range, a minimum flow, or a preferred maximum flow. As with hydraulic methods, the minimum flow can be defined as the breakpoint or as the flow at which the habitat has dropped to a certain percentage of its value at mean or median flow. It can also be defined as the flow that has the lowest acceptable minimum amount of habitat in absolute terms. If minimum flows are at or above maximum habitat for a particular species or instream use, the area of habitat available to that species will be less than maximum for most of the time. Often this does not matter because the rate of change in habitat with flow is less at high flow than at low flow (Fig.5.2) and the difference between maximum habitat and the amount of habitat at a high flow is relatively small. For example, most New Zealand native fish are found in shallow water along the edges of large rivers (Jowett & Richardson 1995) and there is usually some edge habitat available over a large range of flows. However, if maximum habitat for all species and instream uses is less than the minimum flow, it suggests that a reduction in flow might enhance those values.

When many fish species and life stages are present in a river, there are usually conflicting flow requirements. For example, young trout are found in water with low velocities, and adult trout are found in deep water with higher velocities. If the river has a large natural morphological variation with pools, runs and riffles, some of the different requirements may be provided for. Still, even in these rivers, and especially in rivers with small habitat variation, one species may benefit greatly from a reduction in depth and velocity, whereas habitat for another species will be reduced. If a river is to provide both rearing and adult trout habitat, there must be a compromise. One such compromise is to vary flows with the seasonal life stage requirements of spawning, rearing, and adult habitat, with the optimum flow gradually increasing as the fish grow and their food and velocity requirements increase. Biological flow requirements may be less in winter than summer because metabolic rates and food requirements reduce with water temperature, at least for salmonids (Chapman 1966; Cunjak & Power 1986). A fish's swimming ability and its critical holding velocity are markedly reduced at low water temperatures (Sandström 1983; Rimmer et al. 1985; Heggenes and Traaen 1988; Heggenes et al. 1993; Bodensteiner and Lewis 1994). Some evidence has been found for reduced condition of trout in winter associated with reduced invertebrate food supplies (Filbert & Hawkins 1995; Simpkins & Hubert 2000). If flow requirements of individual species are different, solutions may be found by choosing one with intermediate requirements (Jowett & Richardson 1995) or by defining flow requirements for aquatic communities.

Minimum flow assessments based on hydraulic habitat have been used in New Zealand for 25 years and in that time there have been considerable improvements to the survey and analysis techniques and to our knowledge of habitat preferences of New Zealand aquatic fauna.

Minimum flow assessments using hydraulic habitat are carried out in three steps:

- habitat suitability
- survey and analysis
- interpretation.

The following sections describe habitat suitability models, some of the techniques of hydraulic habitat modelling, and lastly some of the considerations and difficulties in interpreting habitat analyses and applying them to minimum flow assessments. Habitat analysis is an aid in the process of deciding on appropriate flow rules for river management, but it is not a decision making process in itself.

6. Habitat suitability models

Key points about habitat suitability

- Depth, velocity and instream cover are basic requirements.
- The simple nature of habitat suitability criteria, when presented graphically, belies the effort behind their development.
- Sample a wide range of habitats with equal effort if possible.
- Collect as much data on habitat suitability from as many rivers as possible, and revise criteria as more data become available.
- Habitat preference is based on a comparison of habitat in which the species is found and habitats available in the river, but interpret the results carefully.
- Consider the life stage or activity with the highest flow requirement and don't confuse activities (e.g., resting and feeding).
- Alternative habitat models, such as fish bioenergetic models and generalised additive models, provide a means of independently checking conventional habitat models.

The concept of good habitat is familiar to most people. For example, angling texts from the turn of the century describe likely trout streams and more recent books (e.g. Hill & Marshall 1985) accurately describe locations where trout are likely to be found. It is possible to determine the relative quality of the different habitats from the abundance of animals in them. Usually, animals are most abundant where the habitat quality is best, in lesser numbers where the habitat is poor, and absent from totally unsuitable habitat. In the aquatic environment, instream habitat usually refers to the physical habitat – water velocity, depth, substrate, and perhaps cover.

Habitat criteria have more influence on flow assessments than any other aspect of the analysis. Failure to use appropriate criteria can result in inappropriate flow assessments and this is one reason that the use of habitat suitability criteria has been criticised. Therefore, habitat criteria need to consider all life stages and, where appropriate, include suitability criteria for the production of food for those life stages. Selection of appropriate criteria and determination of habitat requirements for an appropriate flow regime requires a good understanding of the species' life cycles and food requirements (Heggenes 1988; 1996).

Flow assessments based on habitat (IFIM) have been criticised for considering only a target species or a limited number of species and ignoring biotic interactions. However, most habitat suitability criteria describe meso-scale habitats in which the organisms are found. This contradicts the widely held belief that habitat suitability criteria usually define microhabitat. Measurements of habitat use are taken at a fish's position where the depth and mean column velocity measurements will be similar to those in the general vicinity. Nose velocities (velocity taken at a fish's nose) fall more strictly into the category of microhabitat measurements, but generally these are not used in traditional habitat modelling. Hydraulic models (other than 3D models) predict mean column velocity and so this is the variable used to determine velocity suitability. Although the use of nose velocity seems reasonable from a biological and energetics point of view, fish can select an appropriate nose velocity by moving up or down in the water column. The range of mean column velocities used by fish encompasses the range of possible combinations of depth and nose velocity that they use. This is evident in data collected by Hayes & Jowett (1994), where the range of mean velocities used by adult brown trout was greater than the range of nose velocities. Moreover, habitat suitability criteria for New Zealand native fishes typically have been based on measurements in lanes or areas of 2–3 m^2 of 'homogeneous' depths and velocities from which the species have been collected by electrofishing (Jowett & Richardson 1990, 1995). Benthic invertebrate suitability is based on benthic invertebrate densities measured in sample areas of 0.1 m^2. The nose or microhabitat velocity for benthic species is zero or very near zero and clearly these measurements relate to mesohabitat rather than microhabitat. The mesohabitat types (e.g., pool, run and riffles) used in habitat mapping (Section 7.1.1) are larger units of area and more varied hydraulically than the habitat use mesohabitat sampling units.

Mesohabitats are also occupied by other organisms and may therefore include biotic interactions. For example, good riffle habitat provides for a number of native fish species, juvenile trout and benthic invertebrates. Habitat suitability criteria that describe the locations in which these organisms are found also describe what could be termed 'good riffle habitat'. Thus, habitat requirements of species can be used as indicators or surrogates for broader mesohabitat values. For example, the habitat suitability criteria for common and redfin bullies specify optimum depths of 0.11–0.19 m and velocities of 0.28–0.38 m/s. As a rule of thumb, good run habitat occurs where the numerical value of the velocity exceeds 1.24 times the numerical value of the depth, so the common and redfin bully criteria describe habitat that is intermediate between run and riffle. Biotic interactions that affect habitat use, such as competition for space or reaction to predation, can be modelled by the use of appropriate suitability criteria, but first the existence of an interaction needs to be established. Bonnett & McIntosh (2004) found that juvenile trout had no effect on habitat selection by inanga, whereas Baker et al. (2003) found that flathead galaxias were found mainly in riffles when trout were present, but used a wider range of habitats where trout were absent. Although there have only been limited studies of effects of interactions on habitat use, there is clear evidence that abundance and distribution of native fish can be affected by the presence of trout (McIntosh et al. 1994; McDowall 2006).

6.1 Habitat preference and suitability curves

Many aquatic species are found in similar hydraulic conditions in a wide range of rivers. Their locations are found by electro-fishing for small benthic fish, bank and snorkel observation and high resolution sonar (DIDSON) for large trout, spotlighting for nocturnal habitat use, or Surber sampling for invertebrates. The hydraulic conditions at sampling points are measured and summarised in habitat suitability curves, which are indices of the frequency or abundance with which the aquatic species are found in the particular habitats.

The concept of habitat suitability was applied in biological studies before the development of instream habitat modelling. In one of the first New Zealand studies of habitat suitability, Campbell & Scott (1984) found that 0+ brown trout moved from runs to pools and adopted shoaling behaviour when water velocity in runs fell below 0.3 m/s. They suggested that run habitat was optimum for juvenile brown trout and that 0.3 m/s could be used as a minimum velocity criterion. Now, New Zealand habitat preference curves have been developed for adult brown trout, adult and juvenile rainbow trout (from a limited set of rivers), juvenile Chinook salmon (but from only one river), 12 common benthic invertebrate species or groups, 14 native fish species, and salmon angling (Glova & Duncan 1985; Jowett et al. 1991; Hayes & Jowett 1994; Jowett & Richardson 1995; Hayes & Strickland 2002). Additional data on habitat use have been collected since these studies and a full description of these will be published in a separate NIWA Technical Report.

Habitat preference data for native fish (Jowett & Richardson 1995) and common benthic invertebrate species or groups (Jowett et al. 1991; Jowett 2000) were based on measurements of insect or fish densities within small habitat units, unlike trout habitat measurements where the characteristics of individual fish locations were measured. Thus, there is an implied relationship between fish or insect density and habitat that requires little validation. However, comparisons of native fish density and instream habitat in a sub-catchment of the Grey River (Jowett et al. 1996a) showed that fish densities were generally highest where the average stream depth and velocity were within the range of preferred depth and velocity. Unlike brown trout, native fish abundance was not directly related to benthic invertebrate abundance.

Habitat suitability curves have also been developed for threatened species (e.g., blue duck; Collier & Wakelin 1995) and recreational activities (Mosley 1983; Hayes & Strickland 2002).

The terminology surrounding habitat suitability can be confusing. Here, we define habitat use by the frequency of counts or abundance of aquatic organisms in a specific habitat, such as a range of water depth or velocity. Habitat preference is usually calculated by dividing the frequency of habitat use by the frequency with which habitat is available and then normalising to a maximum value of 1. Bovee (1986) described three categories of habitat suitability criteria. Category I curves are based on expert opinion. Category II curves are based on the frequency of habitat use, and Category III curves are based on preference by adjusting habitat use for habitat availability as described above. Thus, frequency of use curves are equivalent to Bovee’s Category II curves and preference curves are equivalent to Category III curves. We use the term habitat suitability to refer to curves developed from a subjective interpretation of habitat use and preference, where habitat use and availability data are collected in a wide range of stream conditions in order to define optimal habitat and suitability criteria for broad-scale application (e.g., Jowett 2002). Factors, such as swimming ability and bio-energetic requirements, should also be taken into consideration when considering habitat suitability criteria.

It is very difficult to determine habitat suitability without bias because study river(s) invariably provide a limited range of habitat and there is always a tendency for the sampler to introduce bias by concentrating on locations where the likelihood of catching fish is greatest. Data on where the fish are not present is just as important as data on fish presence. In addition, factors other than physical habitat alone may influence fish behaviour – e.g.:

- food
- predation risk
- competition
- physiological and hunger state
- temperature
- light / time of day
- turbidity.

6.1.1 Calculation of habitat suitability

The simplest form of habitat use data is a series of measurements of the habitat in which the target species or activity was observed. These measurements are analysed to determine the habitats that are most likely to used. These data form the basis of habitat suitability criteria.

Habitat suitability criteria can be derived independently for water depth, velocity and substrate composition, assuming that habitat use and availability were sampled randomly throughout each of the study areas (sampling protocol C; Bovee et al. 1998). The most familiar index of selection (or preference); at least with respect to habitat suitability criteria, is the forage ratio, which expressed in terms of abundance is:

$$w = \left. \frac{u_i}{\sum u_i} \middle/ \frac{a_i}{\sum a_i} \right.$$

where w is the forage ratio, u_i is the total abundance/number in habitat units of category i (for example, velocity between 0.2 and 0.25 m/s), Σu_i total abundance/number over all habitat unit categories, a_i is the area of habitat units of category i available in the sample and Σa_i total area of habitat units sampled (Manly et al. 1993). Thus, the forage ratio is the average abundance in each habitat divided by the average abundance overall.

This discrete formulation depends on the 'width' of sampling unit and rarely forms a smooth function. However, the forage ratio can be expressed as a continuous function in the habitat unit by dividing the frequency of abundance by the frequency of habitat availability, where frequency functions were derived by kernel smoothing (Hayes & Jowett 1994; Jowett 2002), and this method has been used to derive habitat preference in most New Zealand studies.

Suitability indices (*si*) are derived from the forage ratios by dividing the forage ratio by its maximum value, so that the suitability index has values of between 0 and 1.

The sampling methods and strategy used to obtain data for habitat suitability studies need careful consideration. The method of sampling should not disturb the fish, the selection of sampling sites should be unbiased, and a large range of habitats should be sampled. For example, if the habitat preferences of a fish species are known or suspected, there may be a tendency to sample only those habitats likely to contain those fish. If sampling is biased towards fish locations, there will be relatively little variation in number of fish found over the range of habitats sampled.

Ideally, a sampling programme is designed to sample a wide range of habitats with near equal effort. The preference calculation is an attempt to counter habitat availability bias caused by the full range of habitats not being sampled with equal effort – but it can itself introduce bias/distortion, especially in regions of low frequencies in either the habitat use or availability distributions (e.g., commonly at the tails of the distributions).

No adjustment for availability is necessary for measurements of density (abundance per unit area), because it is assumed that fish/insects etc. will be most common where the habitat is best. If a range of habitats is sampled, the average density per sample in each interval range is a measure of habitat suitability. Standardisation of density data may be necessary if they are collected at different times or in different rivers. Standardisation converts actual abundance to relative density in each river, or group. An alternative, and possibly better, approach is to develop preference curves for each river and then average those curves so that equal weight is given to each river, irrespective of density.

As with habitat use data, availability data should be collected in all habitats available in the river and must be collected without bias. Available habitat data can be analysed as counts to determine the frequency with which a habitat variable occurred within the river. Instream habitat survey data can be used to estimate the habitat available in the river. Measurements of habitat are weighted by the area represented by each data measurement to determine the frequency distribution of available habitat.

6.1.2 Adjustment of habitat use for habitat availability

The adjustment of habitat use for habitat availability (i.e., calculation of forage ratio) is usually the preferred method of calculating habitat preference, when the data consist of observations of habitat use in a single river and where the range of available habitats may be restricted or at least biased.

As described above, the availability observations are used to derive the proportion of each habitat range in the river $a_i / \sum a_i$ and the habitat use counts are adjusted accordingly.

The calculation of preference is subject to uncertainty when sample sizes are small. The division of a small number by an even smaller number can give a spuriously high preference. In Figure 6.1, the secondary peak in preferred velocity (0.5 m/s) is spurious and is a result of low availability at velocities greater than 0.5 m/s.

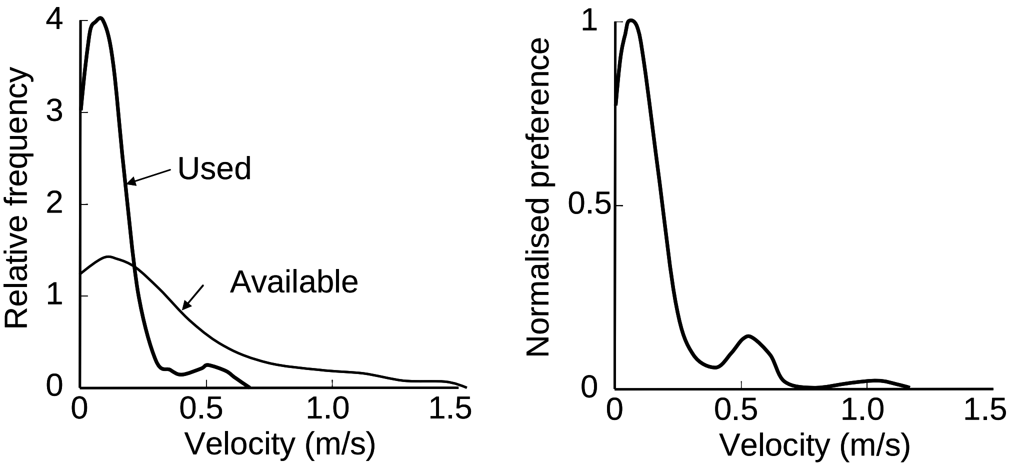

Figure 6.1: **Derivation of velocity preference curves for juvenile rainbow trout from comparison of the locations used by fish (left) and available habitat (left) to derive preference (right). Note how the occurrence of a few fish at higher velocities is exaggerated by the preference calculation.**

Final determinations of habitat suitability are often made subjectively after considering both habitat use and average fish abundance, giving less weight to parts of the curve where the preference is based on a small number of samples (e.g., the low sampling frequency at velocities above 0.5 m/s in Fig. 6.1). Preference curves are sometimes simplified for use in computer analysis. For example, an optimum range of velocities might be specified rather than the single optimum velocity that results from the numerical calculation. A typical modification might assign a habitat suitability value of 1 to preference values of greater than 0.8 and preference values less than 0.2 could be assigned a suitability of zero. There is no analytical reason for this simplification, but it does seem reasonable that a range of values (e.g., velocities) can provide optimum conditions. Replacing low preference values with a zero avoids the possibility that the WUA is made up of a large area of marginally suitable habitat. However, there are no hard and fast rules, and we have found that such modifications have little effect on the assessment of flow requirements.

6.1.3 Standardisation of data

If abundance (density) data is collected in a number of different locations where the population densities may be affected by factors other than habitat, the data can, and should, be standardised between locations, or groups, (e.g., by river) to give the abundance at each sampling location relative to the average (or maximum) for that group (river).

Abundance data can be standardised (i.e., divided by the mean or maximum) so that suitability curves are in terms of mean or maximum values in the dataset. If standardised by the maximum, relative abundance is always between 0 and 1. If standardised by the average, abundance is relative to the average of the group.

Suitability curves can also be derived for each site independently and then averaged between sites, thus avoiding the need for standardisation. This method has the advantage that it is possible to examine the consistency of habitat preference between locations.

6.2 Trout habitat suitability

Adult brown trout habitat suitability criteria in New Zealand appear to be consistent between rivers. This characteristic is termed ‘transferable’ in some literature. The physical characteristics of drift-feeding locations used by over 400 large brown trout were measured in the Travers and Mataura rivers (Hayes & Jowett 1994). Similar water velocities and depths were utilised in both rivers although the availability of these was different in each river (Fig. 6.2). More recently, adult brown trout locations were measured in the large Clutha River (Fig. 6.3) and these showed that the preferred velocity was 0.5 m/s, practically the same as in the much smaller Travers and Mataura rivers, but that brown trout were found in much deeper water than in the Travers and Mataura. The habitat suitability curve for depth derived from Hayes & Jowett (1994) data (shown in Fig. 6.4) considered that any depth greater than 0.6 m was ideal habitat and this assumption is supported by the Clutha data.

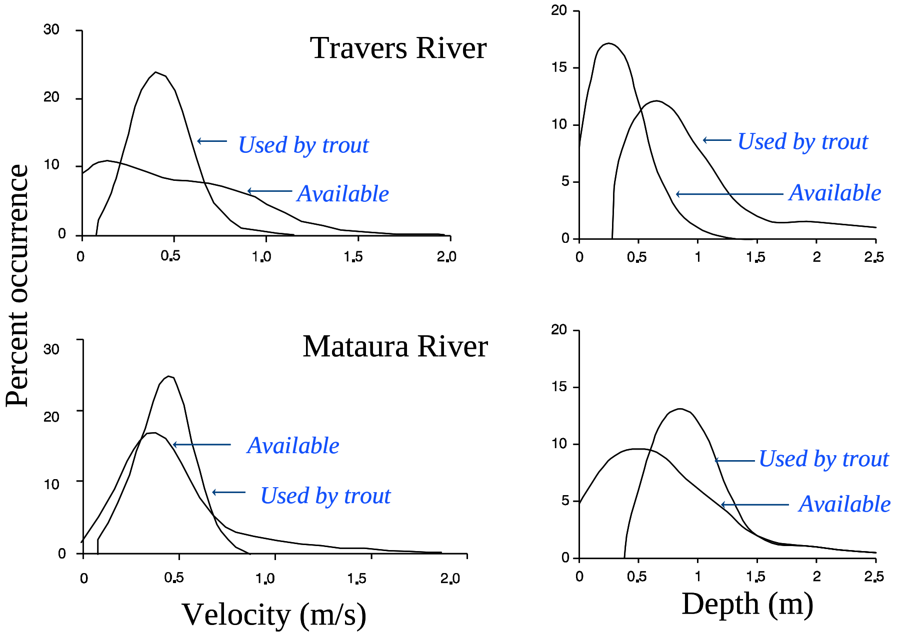

***Figure 6.2*:** **Comparison of frequency distributions of habitat used by large brown trout with the habitat that was available in the rivers.**

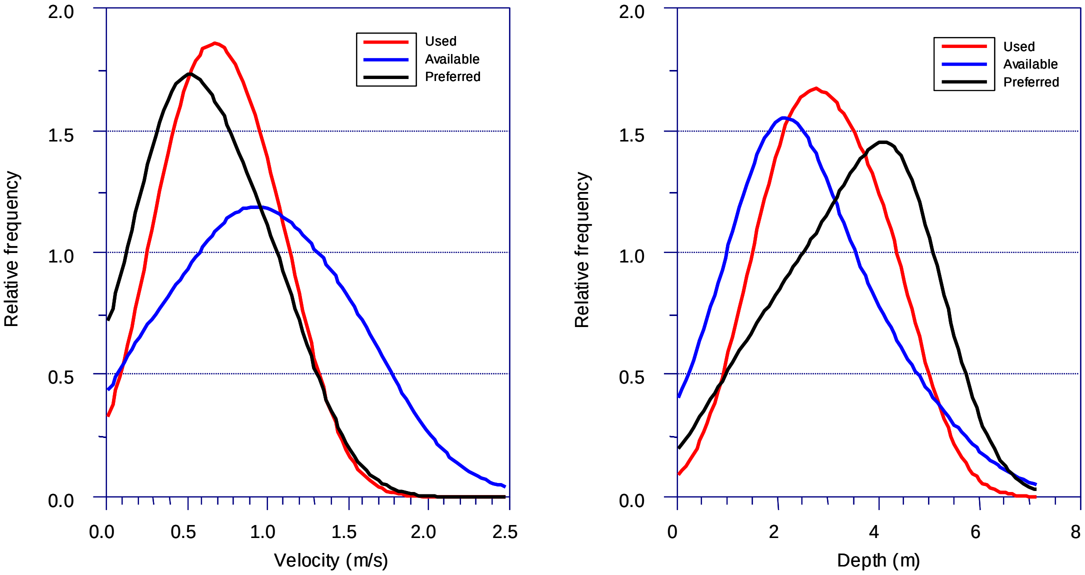

Figure 6.3: **Velocity use, availability, and preference for large brown trout in the Clutha River at the Lake Wanaka outlet.**

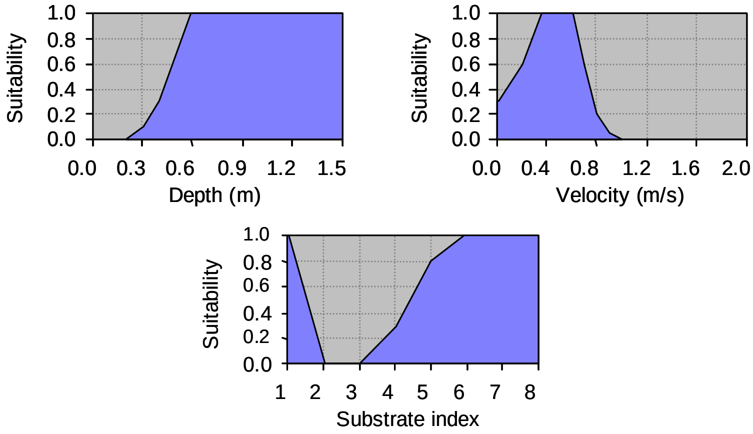

Figure 6.4: **Habitat suitability curves for adult brown trout (adapted from Hayes & Jowett 1994).**

Consistency in habitat use between rivers is not surprising as water velocities occupied by biota are often dictated by the size of the organism, its behaviour and physiology. For example, if a fish feeds in the current its preferred feeding velocity will be related to its swimming ability (Fig. 6.5). Habitat suitability curves for brown and rainbow trout should show an increase in preferred velocity with fish size. Theoretically, velocity use might vary between rivers because of differences in substrate size and water depth and the ability of the fish to find suitable feeding and shelter locations in the available habitat. At present we have insufficient data to investigate this possibility fully.

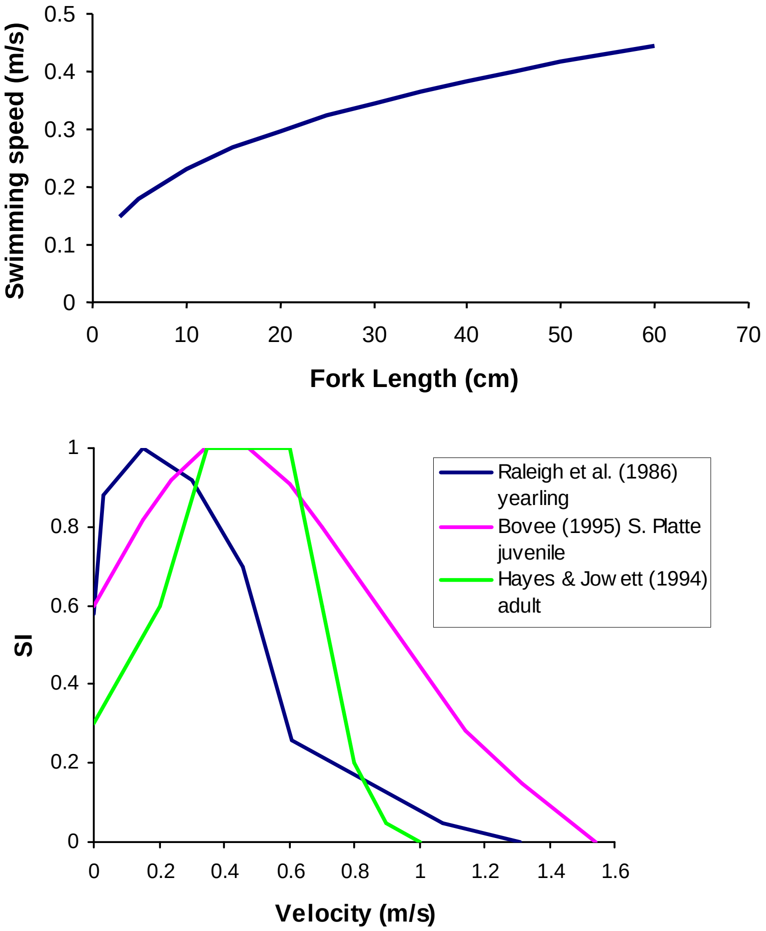

Figure 6.5: **Relationship between fish size and optimal swimming speed for brown trout (above) and habitat suitability curves for juvenile and adult brown trout (below).**

The brown trout suitability curves in Figure 6.4 were developed for New Zealand adult trout (Hayes & Jowett 1994) and specify higher depth and velocities than curves for adult brown trout developed in the U.S. (Raleigh et al. 1986). Whether this is due to differences in the sizes of fish has not been clarified. However, it is clear that it is important to use suitability curves that are appropriate to the river and were developed for the same size and life stage of fish, and behaviour, as those to which they are applied. Raleigh's brown and rainbow trout suitability curves fell out of favour for instream habitat modelling in Colorado after their predictions were found to be unrelated to trout abundance in Colorado Rivers (K. Bovee pers. com.). Suitability curves from the South Platte River (see Thomas & Bovee 1993 for rainbow trout criteria) were developed as replacements.

6.3 Benthic invertebrate habitat suitability

Stream benthic invertebrate densities are influenced by water velocity, depth, and substrate, with some species favouring swift-flowing water and others slow flowing. Benthic invertebrates are less mobile than fish and their distribution and abundance will be influenced by preceding hydraulic conditions (because they take longer to redistribute in response to a change in hydraulic conditions). For example, they will not be present in an area that has been recently inundated and will be unable to live in a varial zone that is frequently wetted and dried. If hydraulic conditions change, benthic invertebrates may be found in high densities in an area that now contains unsuitable habitat, but that had previously been suitable habitat, simply because they have not had sufficient time to move away. Collection of benthic invertebrate samples for derivation of habitat suitability criteria should be undertaken after several weeks of stable flows.

Benthic invertebrate habitat suitability criteria are derived from measurements of invertebrate densities. Thus, habitat suitability is directly related to invertebrate abundance and the total number of an invertebrate species will relate to the total amount of invertebrate habitat. However, some invertebrate species are more closely related to hydraulic habitat than others. Species with high velocity requirements, such as the caddisfly *Coloburiscus humeralis*, are highly correlated with habitat suitability, while others such as the cased caddisfly *Olinga feredayi*, are poorly correlated (Jowett 1992b).

Although habitat suitability criteria have been derived for a number of species, using data from large rivers, it is doubtful whether these criteria are transferable to smaller rivers. Suitability criteria derived from rivers of 10–195 m^3/s mean flow were found to be transferable to the braided Waitaki River with a mean flow of 360 m^3/s (Jowett 2003a). However, water depths in small streams are usually less than the preferred depths in large rivers, but small streams do contain good benthic invertebrate communities. As the name implies, benthic invertebrates probably relate to near-bed conditions more than average conditions in the water column above them, but studies to determine the precise nature of this relationship were inconclusive (Jowett 2003b). Although habitat suitability clearly varies with river size (Fig. 6.6), the hydraulic explanation for the differences between rivers was not evident within rivers (Jowett 2003b). Many benthic invertebrate species are most abundant in riffle habitat, regardless of river size. If so, it might be more appropriate to use habitat suitability criteria that describe typical ‘riffle’ habitat in the size of river being investigated than depth and velocity criteria developed in large rivers. One means of doing this is to use habitat suitability criteria for riffle dwelling fish, such as redfin bullies, as an indicator of stream invertebrate health in flow assessment.

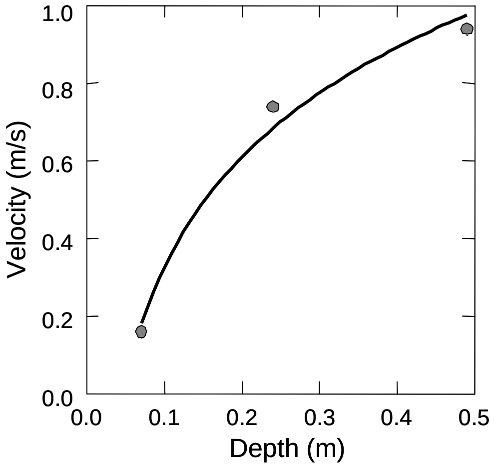

Figure 6.6 : **Logarithmic relationship between mean depth and velocity occupied by *Coloburiscus humeralis* in small (depth 0.1 m), medium (depth 0.25 m), and large (depth 0.5 m) rivers.**

At present, there are insufficient data to define habitat suitability criteria for benthic invertebrates in small to medium-sized streams.

6.4 Composite habitat suitability index

Once habitat suitability curves have been derived for the independent variables, independent suitability index scores for depth (si_d), velocity (si_v) and substrate (si_s) can be calculated for a given point in the habitat survey reach and then multiplied together to form the commonly used composite habitat suitability index (*HSI*) (Jowett et al. 1991; Bovee et al. 1998):

$$HSI = si_d \times si_v \times si_s$$

Other formulations, such as a geometric mean, are possible but multiplying the indices together is the most commonly used method, and has a certain logic because habitat suitability is zero if any one of its components is zero.

6.5 Alternative habitat suitability models

Hydraulic habitat models have been criticised (Mathur et al. 1985; Scott & Shirvell 1987; Castleberry et al. 1996; Kondolf et al. 2000) and much of this criticism has focussed on habitat suitability criteria and interpretation of WUA. Morhardt & Mesick (1988) summarised the criticisms as follows:

1) When calculating the combined suitability index, variables are treated independently and potentially significant interactions between variables are ignored.

2) Weighted usable area, which results from the use of suitability criteria, is an index and cannot be measured directly.

3) Different estimates of weighted usable area can be obtained by using different methods of combining the suitability indices.

4) Weighted usable area combines elements of habitat quantity and habitat quality. A large area of low-quality habitat can produce the same weighted usable area as a small amount of high-quality habitat.

Multivariate statistical models, such as exponential polynomials (Gore & Judy 1981; Orth & Maughan 1983; Jowett & Richardson 1990; Hayes & Jowett 1994), quadratic logistic regression (Thielke 1985; Hayes & Jowett 1994), and generalised additive models (Hastie & Tibshirani 1990; see Section 6.5.1) are alternatives for fitting habitat suitability data; they overcome the problem of independence and can incorporate interaction terms.

6.5.1 Generalised additive models

Generalised additive models or GAMs offer a flexible approach to the development of multivariate models that can be used in hydraulic models to predict relative abundance or probability of use. GAMs (Hastie & Tibshirani 1990) have been used in studies of terrestrial ecology to predict the distribution of vegetation types (Leathwick & Rogers 1996; Leathwick & Austin 2001). GAMs combine nonparametric regression and smoothing techniques. Nonparametric regression relaxes the usual assumption of linearity and reveals the shape of the relationship between the independent variables and the dependent variable. Thus, GAMs are well suited to situations where there are multiple independent variables whose effects you want to model non-linearly, and where the dependent variable is not normally distributed. These models can be applied within an instream habitat hydraulic model to predict how probability of occurrence changes with flow, in the same way that habitat suitability criteria are used with a hydraulic model to predict how WUA changes with flow. The models permit the response probability distribution to be any member of the exponential family of distributions, but those that are most likely to be applied to instream flow assessments are non-parametric logistic models, using presence/absence data, and non-parametric log-linear Poisson models, using abundance data.

This provides an alternative approach to the development and application of habitat suitability and removes some of the subjectivity associated with the development of suitability criteria, the restrictions imposed by assumptions of a mathematical form (such as in exponential polynomial relationships), and satisfies some of the criticisms of independent habitat suitability criteria. Specifically,

- variables are not treated independently,
- interactions between variables can be considered, and
- predictions, such as probability of occurrence, are measurable.

A GAM model based on depth (d), velocity (v), and substrate (s) can take the form:

$prediction = constant + f(d) + f(v) + f(s) + f(sv) + f(sd) + f(dv)$

where each function (e.g. $f(d)$) has a linear and non-linear component fitted by cubic splines and the prediction is transformed into abundance using a reverse logarithmic transform, or to probability of occurrence using a reverse logistic transform. The degrees of freedom are constrained to give a smooth, but flexible, curve. Bovee et al. (1998) note that habitat selection by fish often appears to have thresholds, such as cases where a fish species uses a wide range of depths once the depth has exceeded a threshold. Increasing the degrees of freedom allows the function to adopt a shape that reflects these thresholds. Parameters can be excluded where coefficients are not statistically significant.

Conventional habitat suitability models assign a suitability of 1 to a point where the habitat values are considered optimum. Thus, when habitat suitability values are multiplied by the area they represent and are summed, the resulting number is termed the weighted usable area or area of suitable habitat. However with logistic GAMs, the probability of occurrence is calculated at each point and is then multiplied by the area it represents, before it is summed over the reach. In most cases, the probability of occurrence predicted by a logistic model will be considerably less than 1 and thus the equivalent of ‘weighted usable area’ is a weighted probability of occurrence.

The following example derives habitat suitability models for adult rainbow trout in the Clutha River at the Lake Wanaka outlet by conventional independent analysis and as GAMs. Trout locations were observed by divers and a bank observer in habitats ranging from slow run to rapid. Water depths and velocities at those locations were recorded with a boat-mounted acoustic Doppler current profiler (ADCP). A total of 104 large (> 40 cm) rainbow trout were observed in an average velocity (± std. dev.) of 0.91 ± 0.40 m/s and an average depth of 2.95 ± 1.17 m.

Instream habitat availability data were collected across cross-sections at an average of 2.9 m intervals at a flow of about 170 m^3/s and these were used to predict depths and velocities at a flow of 226 m^3/s; the flow at which the habitat use data were collected. Additional availability cross-section data were collected at 226 m^3/s using an ADCP. These data were restricted to depths greater than about 0.9 m, thus biasing the available habitat dataset towards deep and swift water. Sensitivity tests with and without the ADCP data showed that the suitability models were not strongly influenced by the bias.

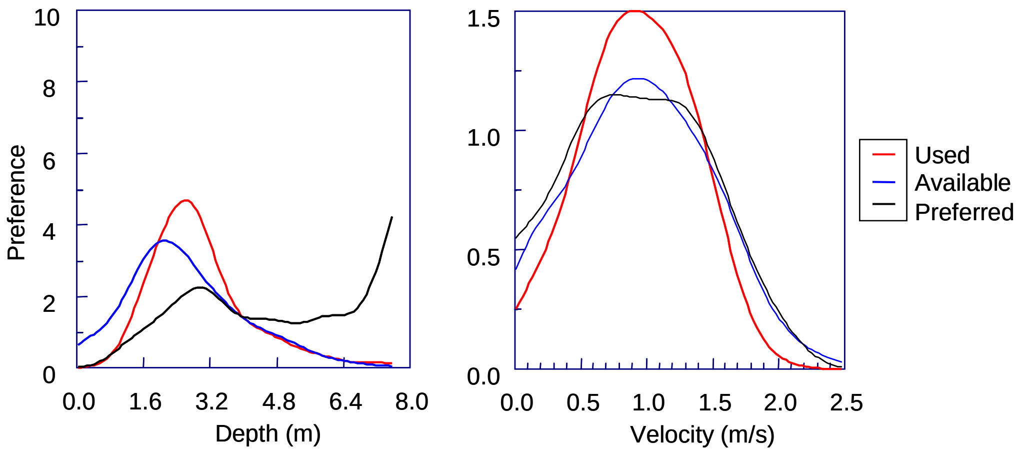

Figure 6.7: **Depth and velocity use, availability, and preference for large (> 40 cm) rainbow brown trout in the Clutha River at the Lake Wanaka outlet.**

Velocity suitability curves (Fig. 6.7) showed that velocities of 0.5–1.4 m/s were preferred by large rainbow trout. Depths of greater than about 3 m were preferred and there was a spuriously high preference for water of about 7 m deep (Fig. 6.7).

The logistic GAMs model developed from the same data (Fig. 6.8), gave similar results to the conventional analysis, showing a slight decline in depth influence when depths exceeded 3 m, and a decline in velocity influence when velocities exceeded 1.4 m/s. The interaction term between depth and velocity was not significant. However, the GAM showed that velocity contribution was high for all velocities between zero and about 1.4 m/s, whereas with the independent velocity suitability curve velocity suitability declined below about 0.5 m/s. The reason for this apparent difference is that the GAM is multivariate and has taken both depth and velocity into account. The apparent decline in trout occurrence at low velocities in the traditional analysis is the result of collinearity. In this river, some rainbow trout were found in deep water with low velocities, but not in shallow water with low water velocities. The GAM was able to account for low probability of occurrence in shallow low velocity water using depth rather than velocity. This demonstrates one advantage of the GAM over independent suitability curves.

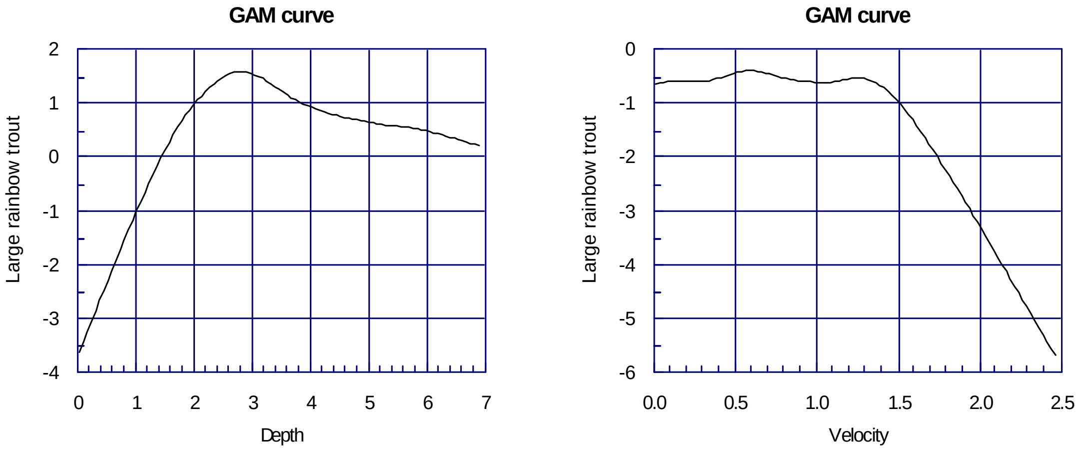

Figure 6.8: **Generalised additive model for large rainbow trout in the Clutha River at the Lake Wanaka outlet.**

When the conventional habitat suitability model and GAM were applied to the instream habitat model of the Clutha River, the shape of the habitat flow relationships were practically identical (Fig. 6.9).

The values of WUA predicted by the conventional model were about 5 times higher than the area-weighted probability of use predicted by the GAM (hence the predicted values have been standardized to a maximum value of 1 in Fig. 6.9). However, as discussed elsewhere it is the shape of the curve, rather than the magnitude, that is important for the assessment of the effect of flow changes on instream habitat.

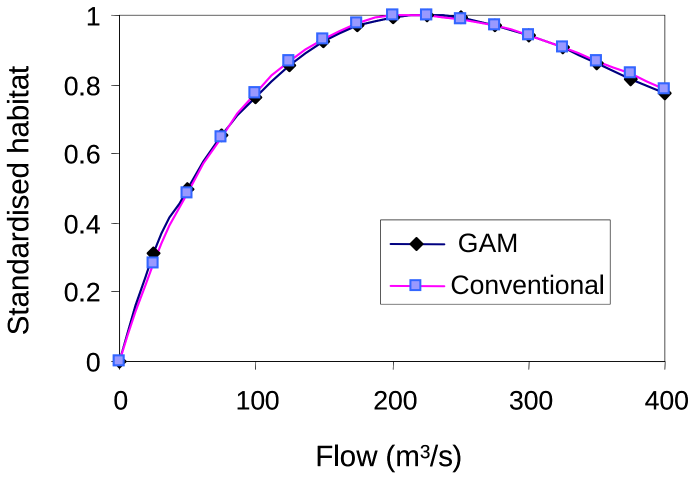

Figure 6.9: **Comparison of habitat prediction for large rainbow trout in the Clutha River at Lake Wanaka outlet using a conventional habitat suitability model (Fig. 6.7) and a generalised additive model (Fig. 6.8). Habitat units have been standardised by dividing by the maximum value.**

6.6 Fish models

Fish models have been developed for drift feeding salmonids and some are at a stage where they can be used for flow assessment. They include models of salmonid behavioural carrying capacity (Morhardt & Mesick 1988), individual-based fish models (Railsback & Dixon 2003) and models based on energetic concepts Addley 1993, 2006; Guensch et al. 2001; Hayes et al. 2000, 2003, 2007; Kelly et al. 2005).

Fish models use the output of hydraulic models and incorporate habitat features and foraging behaviours. Drift foraging models provide a functional understanding of drift feeding and velocity use (Hughes & Dill 1990; Addley 1993, 2006; Hill & Grossman 1993; Hughes et al. 2003). Interest in these models has been driven by a desire for greater biological realism in model outputs. However, this comes at the expense of greater data and model processing requirements. Because these models are fine-scale they apply to representative reaches at the scale of individual riffle/pool or run/pool units and are more expensive to run. Consequently they have a narrower range of applications. Fish models should be seen as complementing, rather than substituting for, broader scale conventional WUA based modelling (1 D and 2 D) (undertaken at broader spatial scales). For example, fish bioenergetic models predict the locations of feeding fish. Habitat suitability curves for feeding fish are based on measurements of the locations of feeding fish. Thus, one approach is theoretical and the other empirical and there should be agreement between the two methods. One advantage of fish models is that they can or could take territorial requirements into consideration to predict potential trout stocks.

There has been interest by stakeholders in extending this energetics approach to native fish but it is either impractical, or difficult to justify for the near to medium term, for the following reasons: 1/ Flow related foraging models and bioenergetics would first need to be developed from scratch – which would take several years of research; 2/ The most common native fishes that drift feed (inanga and smelt) usually co-occur with trout in streams/rivers that typically are the subject of flow investigations, and the latter, because of their larger size, have greater flow requirements and so would be identified as the critical species (see definition of critical species in Section 4.2); 3/ The remaining species are either benthic feeders, a foraging mode that may be independent of flow, or do drift feed but either are uncommon and/or do not often occur in streams/rivers where flow management is an issue.

Advanced fish models utilise functional drift foraging models to make spatially explicit predictions of net rate of energy intake (NREI) based on outputs from 2 d (Guensch et al. 2001; Addley 2006) or 3 d hydraulic models (Booker et al. 2004) and estimates of invertebrate drift density. Most of these models have assumed uniform drift density, but research has shown that drift is spatially variable and flow dependent (Stark et al. 2002).

The most recent development has been to incorporate variable drift density into an advanced spatially explicit fish NREI modelling process (Hayes et al. 2000, Hayes et al. 2007, Kelly et al. 2005). The process links 2D hydraulic modelling (or representative reach 1D modelling) with invertebrate drift dispersion modelling and salmonid drift foraging behaviour to predict NREI and growth potential, and carrying capacity. Initial testing of the drift model showed that it made good predictions of the spatial distribution of invertebrate drift density throughout a single pool in the Travers River at low flow, after it was calibrated against observed drift density at a higher flow. The NREI model correctly predicted the number of adult brown trout observed at one flow in the same pool, but there was not such a good match between predicted and observed spatial distribution of trout. Further research is needed to adequately validate the NREI model. Notwithstanding this need for further validation, these new models are at a stage where they can be gainfully applied to flow regime assessment investigations, where their predictions can be tested against observed spatial distribution of drift density and trout numbers for at least one flow.

6.6.1 Predicting habitat suitability curves with bioenergetics-based drift foraging models

Bioenergetics-based drift foraging models can be used to make predictions of depth and velocity suitability for various sized fish and various water temperatures, which can then be used in traditional instream habitat modelling to predict WUA. The models are based on the functional relationships between NREI and fish size, water temperature and clarity, prey size, and foraging radius. Because foraging models are based on a functional understanding of why fish select certain velocities and depths, they can help with interpretation and selection of appropriate habitat suitability curves from those developed from empirical data on New Zealand rivers, and elsewhere. For example, velocity and depth habitat suitability curves derived from bioenergetics models compare well with some conventional habitat suitability curves for adult brown trout. An example for adult brown trout velocity suitability based on predictions from the Hughes & Dill (1990) bioenergetics drift foraging model is shown in Figure 6.10.

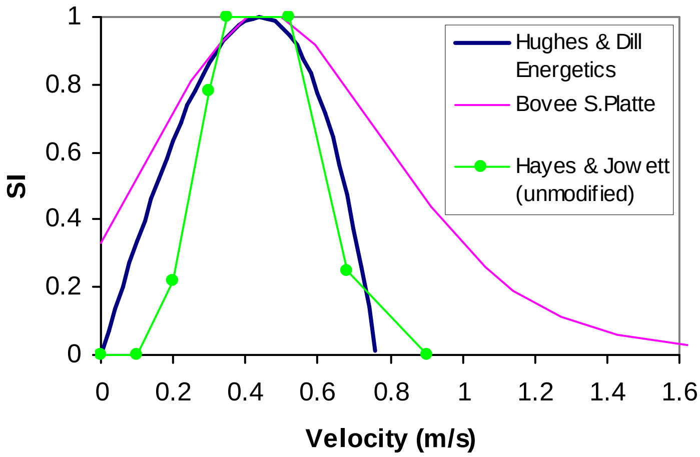

***Figure 6.10*:** **Velocity suitability curves for large drift feeding adult brown trout from New Zealand rivers (Hayes & Jowett 1994), the South Platte River (Thomas & Bovee 1993), and from a bioenergetics model (50 cm trout) (Hughes & Dill 1990).**

Hughes and Dill's foraging model was used to develop hydraulically scaled bioenergetics depth and velocity criteria for a 50 cm trout (Fig. 6.11). Velocity variation through the water column was approximated with a theoretical 1/7th power relationship (Stalnaker et al. 1989).

$V_n = V_m(1.15(Y/D)^{0.143})$

Where V_m is mean column velocity, V_n is velocity at a point in the water column, Y is depth to that point in the water column, and D is the total depth of the water column.

This procedure entailed estimating the foraging radius and mean velocity within a semi-circular foraging area based on the above equation and calculating gross rate of energy intake (GREI) assuming a prey size within the range 5–15 mm (i.e., 10 mm average) and a drift density of 0.4 insects per m^3. These prey sizes and density fall within the range commonly found in New Zealand rivers. The shapes of the GREI x velocity and GREI x depth curves are independent of prey size and density.

Tests of the Hughes & Dill foraging model on large trout in New Zealand rivers indicate that it accurately estimates the foraging area of brown trout (Hughes et al. 2003), but underestimates the foraging area of rainbow trout (N. Hughes & J. Hayes unpublished data). The resulting bioenergetics depth and velocity criteria are likely to closely approximate actual depth and velocity suitability for brown trout, but may underestimate velocity suitability for rainbow trout.

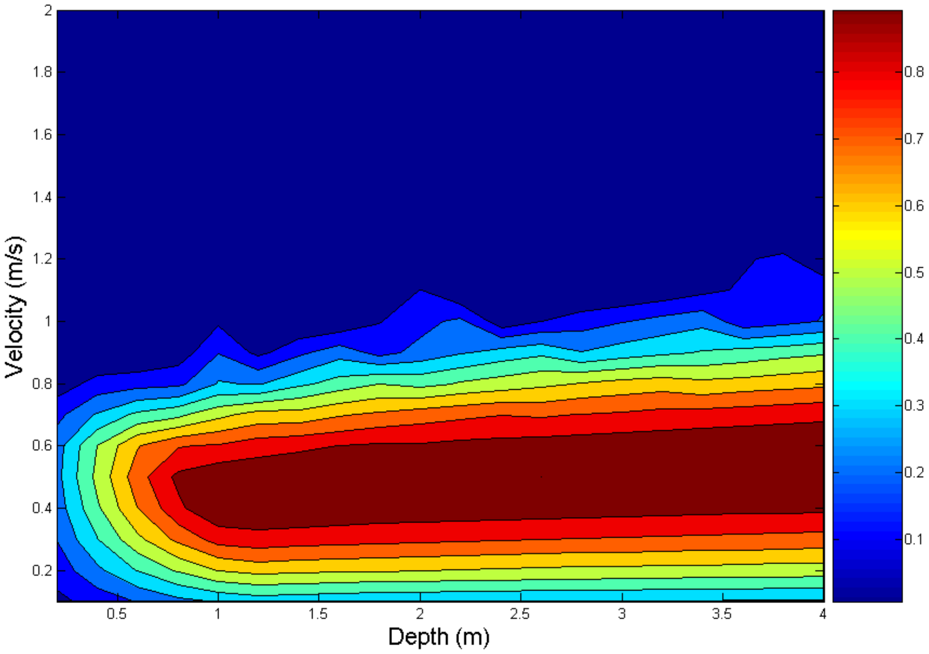

***Figure 6.11*: Gross rate of energy intake (GREI) depth x velocity contour plot predicted from the bioenergetics foraging model for a 50 cm trout feeding on 10 mm drifting prey. Contour colour bar represents magnitude of GREI.**

A generalised additive model was developed for large brown trout observed in the Lake Wanaka outlet of the Clutha River, during the rainbow trout study described previously. These brown and rainbow trout GAMs were used to predict the probability of occurrence in a depth and velocity matrix. These values are shown plotted in Figs. 6.12 and 6.13. The comparison of these graphs shows a high degree of correspondence between the GREI contour plot (Fig. 6.11) and the large brown trout probability of occurrence plot (Fig. 6.12). However, not surprisingly because they are a different species with different behaviours, large rainbow trout do not conform to this trout bioenergetics model, with a higher probability of occurrence at velocities of 1–1.25 m/s than predicted by the bioenergetics foraging model (Fig. 6.13).

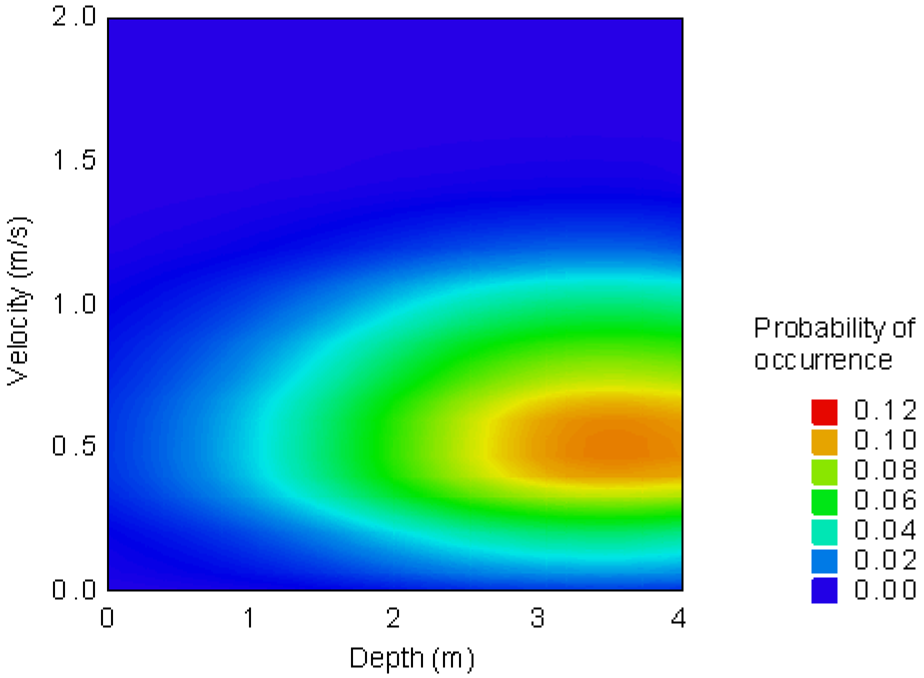

Figure 6.12: **Probability of occurrence by depth x velocity contour plot predicted from the large brown trout Clutha River generalised additive logistic model.**

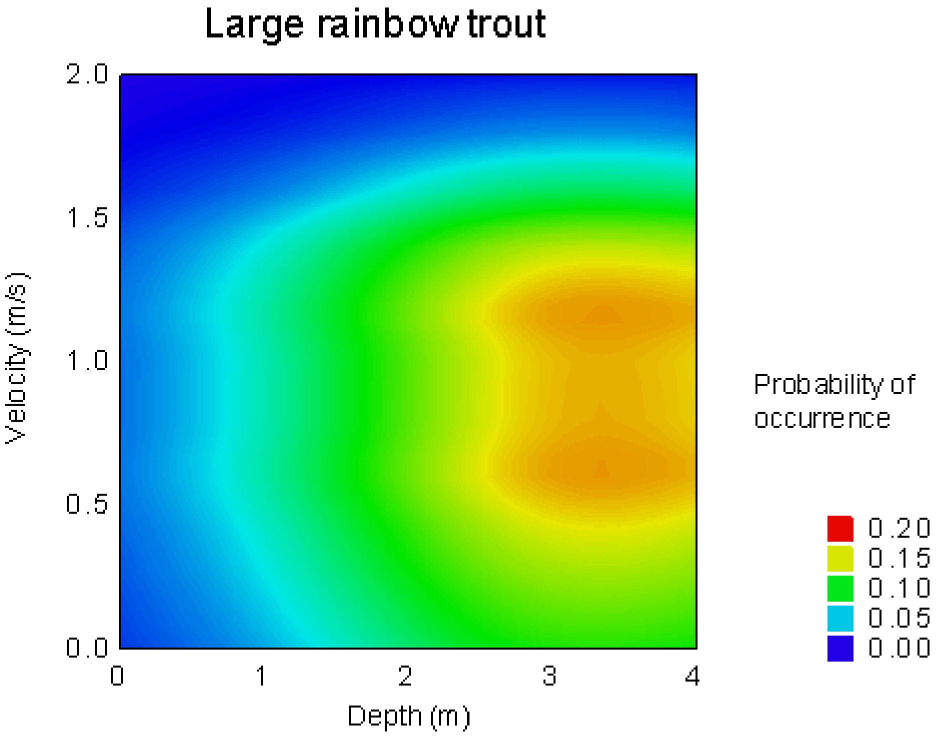

Figure 6.13: **Probability of occurrence by depth x velocity contour plot predicted from the large rainbow trout Clutha River generalised additive logistic model.**

A generalised additive model was fitted to a depth and velocity matrix of GREI values for 10 mm prey. The model fit was good and explained over 98% of the variation in GREI. This model was applied to a surveyed reach on the Clutha River at the Lake Wanaka outlet and the predictions of the variation in mean GREI with flow compared to weighted usable area predictions (HSI) using the Hayes & Jowett (1994) adult brown trout suitability criteria (Fig. 6.4). The two curves for predicted GREI and WUA were very similar in shape and magnitude (Fig. 6.14)

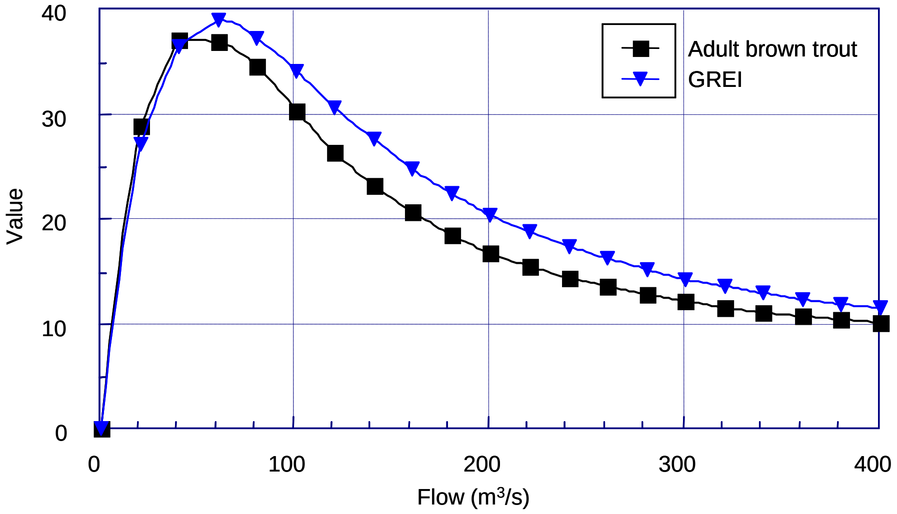

Figure 6.14: **Weighted usable area predicted using adult brown trout habitat suitability criteria (Fig. 6.4) compared to the area weighted sum of gross rate of energy intake (GREI) predicted from depth and velocity using the relationship shown in Fig. 6.11.**

Generalised additive models and bioenergetic models overcome some of the criticisms that have been made of conventional habitat suitability criteria. GAMs provide a more statistically robust way of fitting habitat suitability data, including interaction terms. Bioenergetics based foraging models provide an entirely different approach and produce biologically meaningful metrics GREI and NREI.

Predictions from a bioenergetics based foraging model for brown trout confirmed the empirical habitat suitability curves routinely used in New Zealand for adults of this species (i.e., those based on Hayes & Jowett 1994). However, more research is required on rainbow trout drift foraging and bioenergetics models, before this approach can be confidently used for verification of empirical habitat suitability curves for this species.

7. Instream habitat survey procedures

Key points about instream habitat surveys

- An instream habitat survey describes either average river conditions, or river conditions at critical location.
- To describe average conditions, reach selection should be unbiased, and should represent a longer segment of river where gradient, flow, and degree of bank confinement is similar.
- A representative reach should include at least one morphological (pool/run/riffle) sequence.
- Habitat mapping should cover the range and morphological (mesohabitat) types present.
- Relatively few cross-sections (e.g., 10) or short reach lengths (e.g., 400 m) can adequately determine the shape of habitat/flow relationships.
- Although the amount of habitat may vary between reaches, the shape of the curve is usually similar irrespective of reach selection or survey type.
- Flow assessments are based on the shape of the WUA/flow curve, not the amount of habitat.

7.1 Study area and survey objectives

The objective of an instream habitat survey is to get the best possible representation of the characteristics of a segment of river. This encompasses the range of water velocities and depths that occur in a river, along with the co-occurrence of stationary stream elements (such as substrate, bank formations, and cover) with the hydraulic conditions. It is important that the selection of reaches and cross-section locations should be unbiased and a stratified process of selection is one means of achieving this. In habitat analyses, we use the term ‘section of river’ to denote a long length of river (usually several kilometres or more). A reach is shorter and is usually a kilometre or less. A cross-section or transect is a point within a reach.

The morphology of a river is determined by the strength of banks and bed (riparian vegetation, bank material, and substrate), gradient, and magnitude of flood flows. If any of these factors change, the morphological and hydraulic characteristics of the river will change.

The selection of survey reaches and number of cross-section locations will depend on the river and the issues that are to be addressed. Survey reaches are usually selected to represent the average conditions in a longer section of morphologically similar river. They may also be selected to represent some critical habitat or function, such as a spawning area or fish passage. For example, the shallowest riffles may be modelled to determine the flow at which the depth falls below a critical level for the passage of fish. Surveys of known spawning areas may also be used to determine the effect of flow on spawning habitat.

Rare river conditions or habitats need special consideration. By definition, rare habitats do not occur sufficiently frequently to be included in habitat surveys that aim to describe average conditions. It is possible to carry out a survey of rare habitats and to determine how flow changes would affect them, but before doing this, it is necessary to ascertain:

- the importance of those habitats to the ecosystem,
- whether a change in habitat might create those rare habitats in another location,
- whether those habitats will change with flow or flow regime, and
- what method can be used to predict changes.

Some examples of rare habitats are backwaters, barriers to fish passage, and fish cover requirements in some rivers.

Minimum flow assessments are based on the shape of the WUA – flow curves and the proportional changes resulting from a flow change. Thus, the purpose of the survey is usually to define how habitat varies with flow. The need for a survey to define the shape of the curve is often confused with accurate quantification of the amount of suitable habitat. Fewer cross-sections are required to show the shape of the habitat/flow curve than are required to define the amount of habitat in a reach, as will be shown later. However with few cross-sections, results can be unduly influenced by unusually wide cross-sections, because the characteristics of each cross-section are weighted by the area it represents in the reach.

The number and location of cross-sections in a reach should reflect the variation in morphology and extent of the morphologically similar sections. Significant changes in morphology are usually indicated by a change in gradient, flow, or geology, as noted earlier. However, it is also possible to survey multiple reaches and then combine the hydraulic characteristics and habitat during the analysis to give an average for the river. A river need only be divided into multiple reaches if there is a requirement for comparison of hydraulic/habitat characteristics between reaches. It is often convenient to divide the river into multiple reaches where the flow varies, such as upstream and downstream of a tributary stream. This is because during calibration and analysis of reaches one flow applies to all cross-sections, and it is not necessary to specify the flow at each cross-section.

Reaches can be surveyed in two ways – as representative reach, (usually for water surface profile modelling or 2D modelling) or by habitat mapping (stratified sampling).

7.1.1 Habitat mapping – stratified sampling

Stratified sampling or habitat mapping is used to represent the physical habitat in the segment of river over which the survey is intended to apply, and should provide a better representation of available habitat than simple random selection and will certainly require less effort. In habitat mapping, mesohabitat types with similar hydraulic characteristics are defined and their locations and lengths mapped. Pool, run, riffles are examples of mesohabitat types, but any classification system can be used, as long as it classifies on the basis of hydraulic similarity (primarily depth and velocity). We also use the term ‘mesohabitat’ to describe the hydraulic conditions represented by habitat suitability criteria (Section 6). These habitat suitability ‘mesohabitats’ are sub-areas within the mesohabitat types used for habitat mapping.

Stratified sampling involves two steps:

(a) decide on the number of different mesohabitat types in the segment of river

This involves examining a long section of river and deciding on the number of mesohabitat types with different hydraulic characteristics (i.e., depth and velocity).

(b) divide the segment of river into mesohabitat types (strata)

This involves stratifying the longer section of river into the mesohabitat types, such as pool, run, and riffle, and then selecting cross-section locations in these mesohabitat types.

Stratified sampling first requires that habitat mapping is undertaken over the segment of river under study so that the proportions of the different mesohabitat types (e.g., pool, riffle, run, etc.) can be calculated. To make a habitat map, the entire length or large proportion of the segment of interest is traversed, by foot, raft, or boat. The length of each pool, run, riffle, etc. is measured (by tape measure, GPS, or pacing) and its location recorded.

Next, cross-sections are chosen in each of the mesohabitat types. The cross-sections should be selected without bias, such as would be caused by selecting cross-sections in similar riffles or in similar locations within a pool. The sampling approach generally applied is not random, but targeted to cover the full range of natural variation with only a few cross-sections (rather like estimating the mean by averaging the maximum and minimum value in a data set, rather than just two data points at random). At each cross-section, depths, mean column velocities, and substrate composition are recorded at approximately 0.5–1 m intervals, or with enough frequency to characterise the changes in depth and velocity across the section, exactly the same as for hydraulic modelling. The water level is measured and the flow calculated from the measurements of velocity, depth, and width. To predict how the water level at each cross-section changes with flow, the water level at each cross-section is measured at two, or more, other flows and a stage-discharge relationship established. Water level measurements can be made with survey instruments or read with a ruler off temporary staff gauges (e.g., a length of steel reinforcing bar driven into the river bed). The latter is usually more accurate.

Mapping of a segment of the river is carried out to define the mesohabitat types present and to determine the percentage of each type within the reach. Each cross-section represents the percentage of the mesohabitat type in the reach divided by the number of sections in that mesohabitat type. For example, if riffles made up 25% of a 2 km segment of river and 6 cross-sections were surveyed in riffles then each cross-section would represent 25/6 or 4.2% of the river section.

7.1.2 Representative reach

Hydraulic habitat modelling can be based on ‘representative’ reaches along a river, with closely spaced cross-sections, or topographical measurements, along the reach that are used for 1D water modelling or 2D modelling, respectively. There are two approaches to 1D modelling in representative reaches: 1/ water surface profile modelling, 2/ rating curves. For 1D modelling by either method, water level measurements must be taken at each cross-section at the survey flow. For water surface profile modelling, a stage–discharge relationship must be determined for the downstream cross-section. This relationship is used to predict water levels at the other cross-sections over a range of simulated flows. If rating curves are used, water levels must be measured at all cross-sections for a number of different flows in order to construct stage–discharge relationships for each cross-section. This approach is more accurate, and requires less experience, than water surface profile modelling. Water level measurements are also necessary for 2D modelling – for calibrating and validating the model.

A representative reach contains a range of habitats, usually one or two pool/run/riffle sequences that are considered representative of a longer segment of the river. Closely spaced measurements are necessary to allow accurate interpolation between measured points, both laterally and longitudinally. Closely spaced measurements are particularly necessary to define the channel in transition zones between mesohabitat types (e.g. where a riffle flows into a pool). In representative reach 1D modelling, the distance between cross-sections is usually used to calculate the proportion of the reach that each cross-section represents. The length or reach that each cross-section represents is half the distance between the adjacent upstream and downstream cross-sections. However, it is also possible to specify percentage values for each cross-section, if required.

7.2 Reach selection

Selection of a reach and cross-section locations poses the problem of how ‘representative’ they are of a longer section of river, or even of the hydraulic conditions within the reach. However, experience has shown that although the amount of habitat may vary between reaches, the shape of the habitat/flow relationship is usually similar and neither reach selection nor survey type should affect flow assessments. Superficial differences in appearance of reaches in a river do not necessarily result in differences in the shape of habitat/flow relationships, although they may indicate differences in the amount of available habitat.

The following examples are from the large number of studies, both in New Zealand and overseas, and show that there is remarkably little variation in the shape of habitat curves between reaches, unless a reach is markedly different (i.e. flow, gradient, confinement).

Figure 7.1 shows the results of two habitat surveys carried out on the Arnold River, one reach at Kotuku above the Arnold Dam and the other about 15 km downstream at Kokiri below the Arnold Dam. The Kotuku reach was surveyed as a representative reach and the Kokiri reach was surveyed using habitat mapping. Both curves are of a similar shape, with maximum habitat provided by a flow of 10–15 m^3/s, but the downstream (Kokiri) reach contains almost twice as much usable habitat.

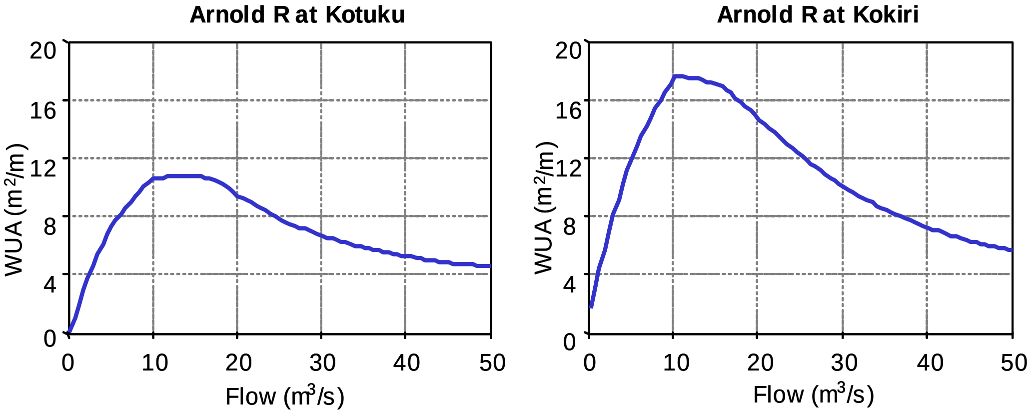

Figure 7.1: **Comparison of instream habitat predictions from two habitat survey reaches of 15 cross-sections about 15 km apart on the Arnold River.**

Figure 7.2 shows predicted instream habitat in three representative reaches in the Esk River, with one pool/run/riffle sequence in each reach (Jowett 1986). With 10–13 cross-sections in each reach, the variation in habitat with flow in each reach was similar for adult brown trout habitat, food producing habitat and common bully habitat. Food producing habitat began to decline sharply when flows fell below 2 m^3/s, adult trout habitat varied little above a flow of about 2 m^3/s, and maximum common bully habitat was at about 1 m^3/s.

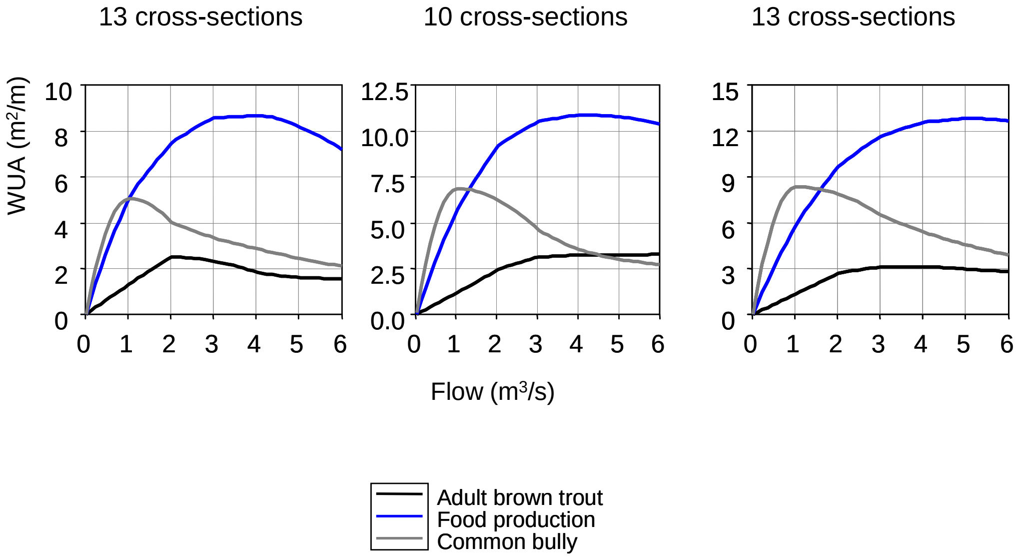

***Figure 7.2*:** **Instream habitat predictions for adult brown trout, food producing, and common bully habitat in three representative reaches in the Esk River.**

In the Kakanui River, the variation of trout habitat with flow showed changes along the river (Fig. 7.3). At the most downstream reach at Pringles, the gradient is low and a flow of about 3 m^3/s is required for maximum adult trout habitat and 1.5-2 m^3/s for spawning. However, further upstream at Riverside the river is more confined and a lesser flow provides maximum habitat. Robbs Crossing is further upstream still, but is poorly confined and flow requirements are more similar to those in the downstream reach. However over the three reaches, a flow of about 1 m^3/s provides close to optimal juvenile trout habitat and flows of 2–3 m^3/s provides good conditions for adult trout.

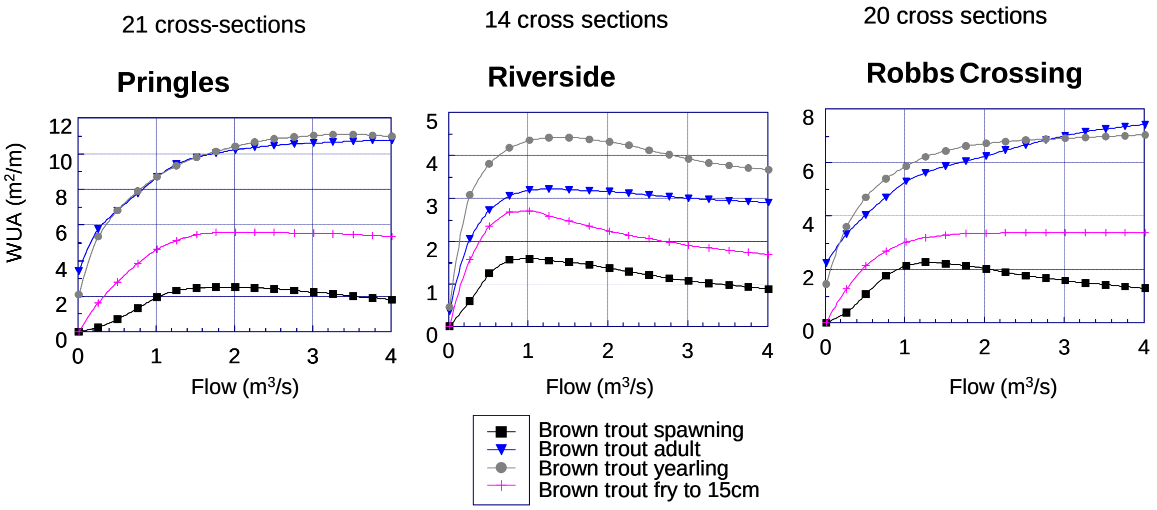

***Figure 7.3*:** **Trout habitat in three reaches of the Kakanui River.**

Two reaches about 40 km apart were surveyed on the lower Waitaki River, a large braided river with a mean flow of about 360 m^3/s. There were 12 transects in the Ferry Road reach (left of Fig. 7.4), with the number of braids varying from 2 to 9 (average 5). The Priests Road reach contained 8 transects, with 4–12 braids (average 7.6). Although the reaches differed in the number of braids, the relationships between habitat and flow were similar in both reaches although the amount of habitat varied by a factor of about 2.

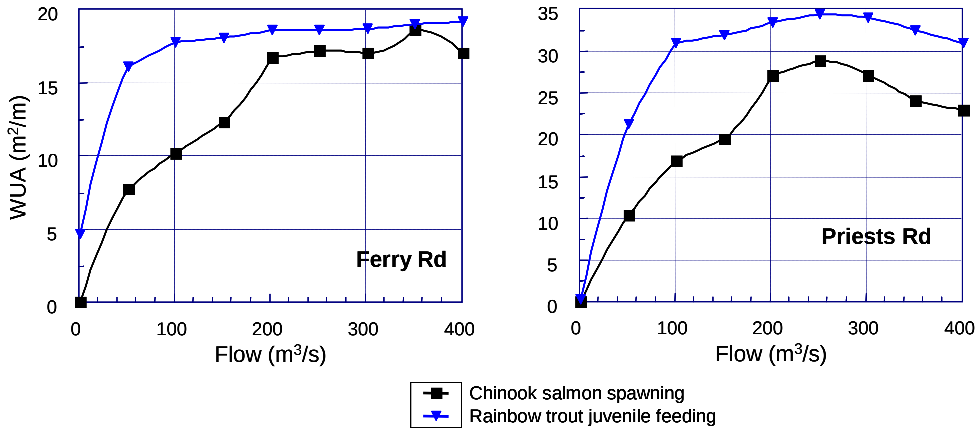

Figure 7.4: **Salmon and trout habitat in two reaches in the large braided Waitaki River.**

These comparisons show that the habitat/flow relationships are relatively consistent within morphologically similar sections of river. Lamouroux & Capra (2002) found that there were also similar habitat-flow relationships between rivers, when the measures of habitat and flow were divided by width. They plotted the habitat suitability index (HSI) against flow per unit width for a number of French rivers and found that the shapes of the habitat curves were similar between rivers, although the magnitude of the HSI values differed between rivers. A similar study showed the same result for over 100 New Zealand rivers (Lamouroux & Jowett 2005). The shapes of the dimensionless HSI/flow relationships were similar between rivers and depended on the habitat suitability curves, but not the river (Fig. 7.5). Thus, each river is not unique in the way water depth and velocity change with flow, even though every river looks different.

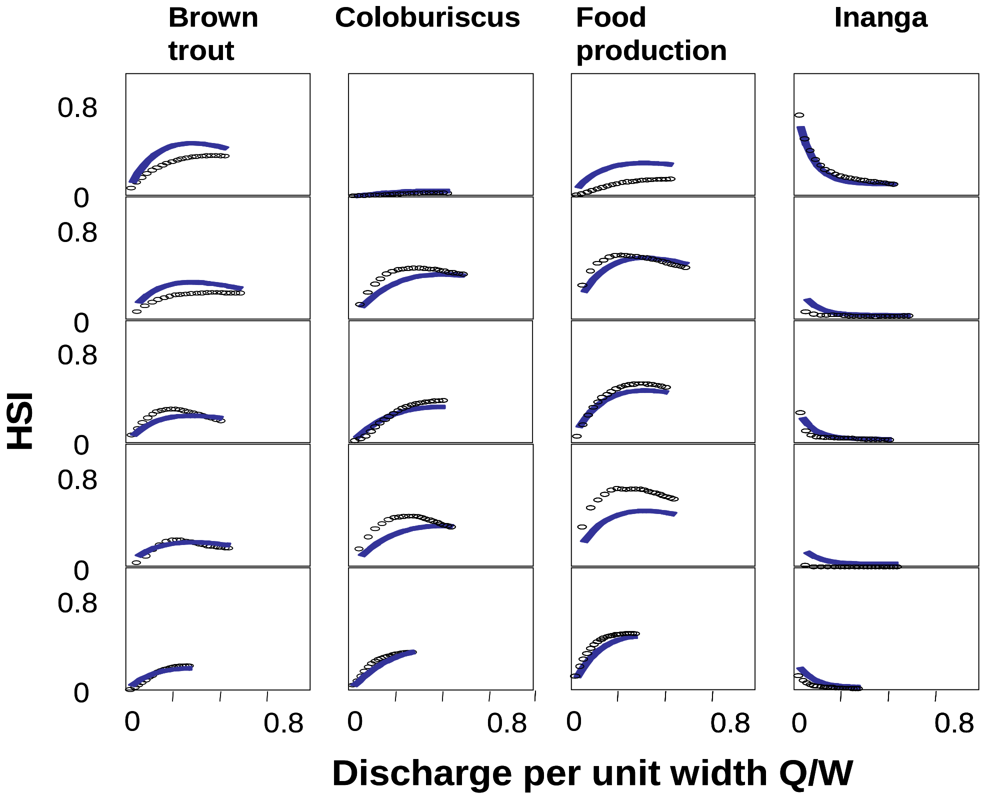

Figure 7.5: **Relationships between habitat suitability index (HSI) and discharge per unit width in 5 rivers. The blue (solid) lines show the generalised relationship and the black points the calculated points for each reach.**

7.2.1 Reach length

In 2D modelling, a representative reach is surveyed and the length of the reach becomes critical because of the amount of data that needs to be collected for an adequate description of the terrain. This is a similar problem to the number of cross-sections in a 1D survey and the answer is the same. Longer reaches give better quantification of the actual amount of habitat in the river, but the shape of habitat/flow relationships can be determined from relatively short reaches.

The following example tests the sensitivity of habitat analyses to reach length by dividing a 1300 m reach of braided river into 2 equal reaches, 3 equal reaches, and 4 equal reaches.

When the 1300 m reach was divided into 2 – 650 m reaches, the shapes of the habitat/flow relationships in each 650 m reach were similar, as well as being similar to the single 1300 m reach, but the amount of habitat in each reach varied, particularly for juvenile brown trout (Fig. 7.6). When the reach was divided into 3 – 430 m reaches, the variation of habitat with flow was similar between each sub-reach and also similar to the single 1300 m reach (Fig. 7.7). However, when the reach was divided into 4 reaches of about 300 m the differences become more noticeable (Fig. 7.8).

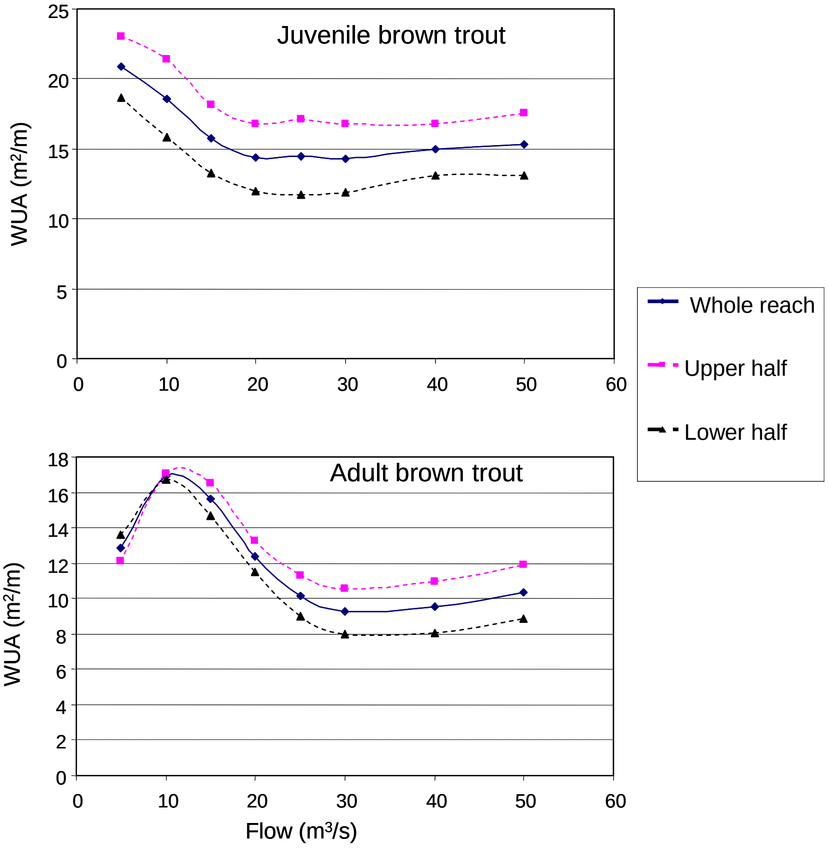

Figure 7.6 : **Relationship between flow and trout habitat in a 1300 m reach of a braided river compared to those in the two halves of the reach.**

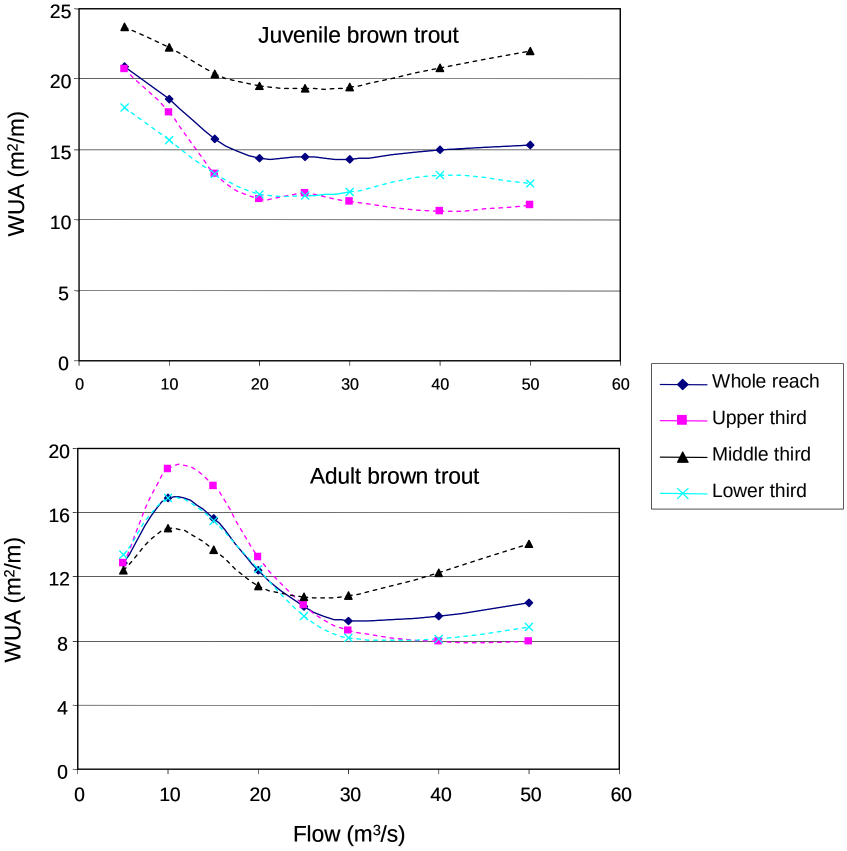

Figure 7.7: **Relationship between flow and trout habitat in a 1300 m reach of a braided river compared to those in the three thirds of the reach.**

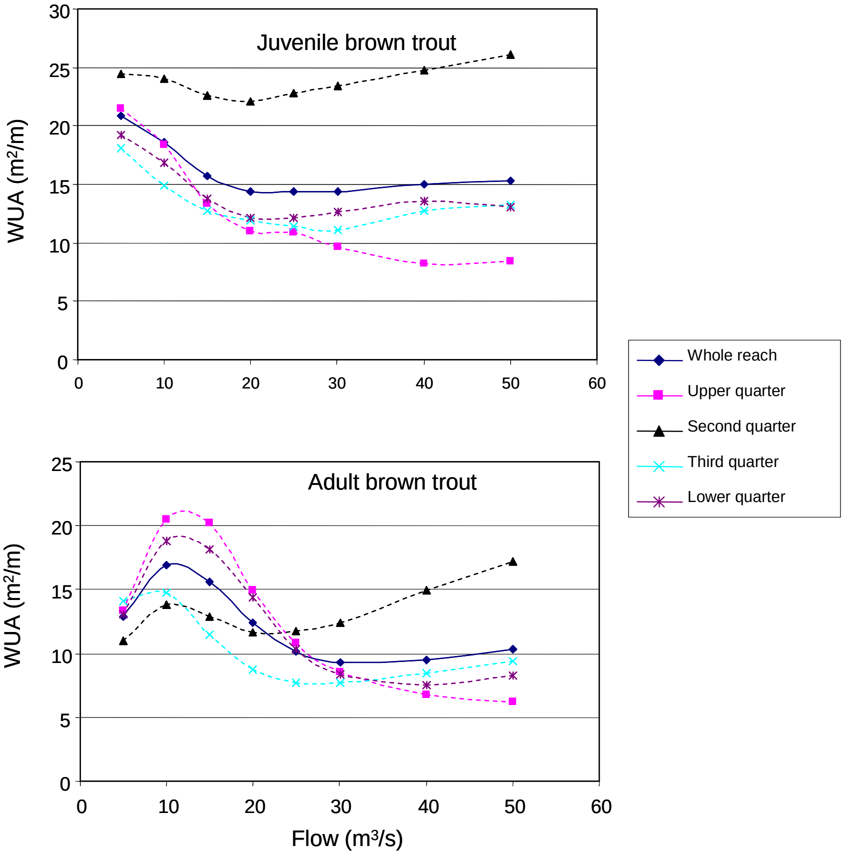

Figure 7.8: **Relationship between flow and trout habitat in a 1300 m reach of a braided river compared to those in the four quarters of the reach.**

7.2.2 Number of cross-sections

The number of cross-sections required in a 1D survey or reach length of a 2D survey depends on the morphological variability within the river. Studies have shown that relatively few cross-sections can reproduce the results from a 2D survey or from a 1D survey with a large number of cross-sections.

- Tarbet & Hardy (1996) found that a 1D survey with 19 cross-sections achieved similar results to a 2D survey.
- Milhous (1990) visually compared results from sub-samples of 4 cross-sections each (one per sampling unit) selected from a set of 24 cross-sections and, with some minor reservations, concluded that 'the shape of the relationships were similar…' and the 'number of cross sections can be relatively small…'
- Simonson et al. (1994) used 86 study sites on 58 Wisconsin streams. They found that 20 transects gave means accurate within 5% of the true mean 95% of the time. With 13 transects, 85% of the means were within 5% of the true means.
- Bovee (1997) concluded that pocket water, a complex mesohabitat type containing a wide variety of depths and velocities, can be accurately described with 3 to 5 transects.
- Payne et al. (2004) sub-sampled several very large data sets to determine how many cross-sections were required to produce a robust WUA function (i.e., habitat/flow relationship), and found that 18–20 cross-sections gave results nearly identical to results for 40 to 80 cross-

sections per reach and only a few cross-sections were required to reproduce the general shape of the relationship (Fig. 7.9).

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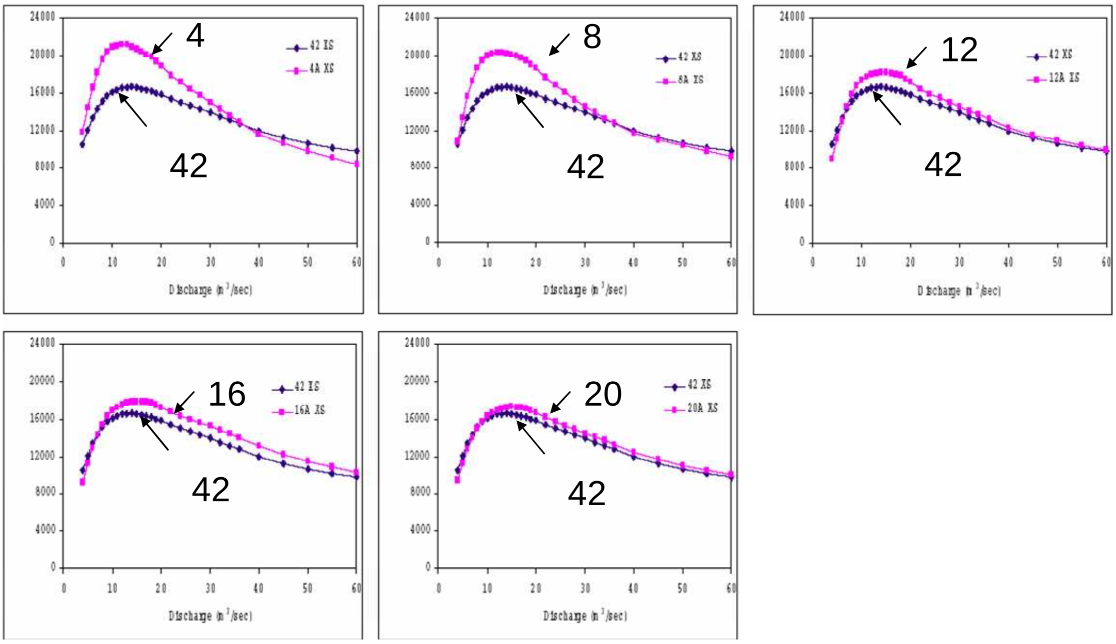


Figure 7.9: **Effect of number of cross-sections on habitat-flow relationships (from Payne et al. 2004).**

The total number of cross-sections needed to generate a robust result should be proportional to the complexity of the habitat hydraulics: 6–10 for simple reaches and 18–20 for diverse reaches. Fewer cross-sections will give the shape of the relationship, but not WUA magnitude.

The number of cross-sections in each mesohabitat type should ensure that no individual cross-section is given a weight of more than 5–10% to minimise the influence of outliers.

- A representative reach should include at least one morphological (pool/run/riffle) sequence.
- Habitat mapping should cover the range of variability within the mesohabitat types present.
- Relatively few cross-sections (c. 10) or short reach lengths (c. 400 m) may be sufficient to determine the shape of habitat/flow relationships, whereas more (20 sections or c. 1200 m) are needed to stabilise the magnitude of WUA predictions.

7.3 Cross-section survey method

The instream habitat surveys are carried out with standard hydrological gauging equipment and can be done by wading, by boat, or raft. In addition, cross-sections need to be marked so that they can be found and identified on return visits and water levels measured.

Usually, a survey aims to provide information on how the habitat or hydraulic conditions vary at low flows. Thus, the survey is best carried out at low flow, to minimise error in extrapolating beyond the measured conditions. However, when flushing and sediment transport flows are of concern, cross-sections should be surveyed to accommodate water levels expected at high flows and calibration measurements should be made at high flows.

When carrying out a stratified survey, the mesohabitat types in the segment of river to be surveyed are determined by examining at least 1-2 km of river. The number of mesohabitat types defined (e.g., pool/run/riffle) depends on the river and survey purpose (see Section 7.1). Once the mesohabitat types are defined, the length of each is measured and cross-sections selected in each mesohabitat type. Often, for convenience of surveying, the first cross-section is chosen in the least common mesohabitat type, with other cross-sections located in adjacent mesohabitat types.

When carrying out a survey of a representative reach, particularly for a water surface profile analysis, cross-sections are located relatively close together, so that there is uniform and relatively small variation in cross-section properties (width, depth, velocity) between cross-sections. Accurate delineation of transition zones, where the water surface gradient is changing rapidly, is particularly important for water surface profile modelling.

Cross-sections should be clearly identified in the field (including the mesohabitat type they represent) and field data (offset distances, depths, number of revolutions and times and especially water levels) should be accurate and systematically recorded.

A tagline or tape is strung across the river at right angles to the flow. It does not matter whether the tape zero is on the left or right bank, but it is preferable to be consistent, so that when plotted data are viewed, cross-sections will be consistently either looking upstream or downstream. The cross-section survey should include points high enough on the banks to accommodate the maximum flow likely to be modelled, and water edges at the time of survey should always be included.

The water level is marked by a temporary staff gauge, such as a reinforcing bar or stake. This is driven into the streambed in a sheltered location on the cross-section in about 10–20 cm of water. Ideally, each temporary staff gauge should be referenced to a benchmark established on the bank so that any movement in the temporary gauge can be detected and corrected if necessary. The water level at each temporary staff gauge is referenced to its top (zero if flush with the water surface). If this method is used, a gauge can accurately measure small changes in water level for derivation of the cross-section rating curve. At each cross-section, stage at zero flow should also be estimated to provide another point for fitting the stage–discharge relationship. The stage at zero flow for riffles is usually the lowest point in the cross-section (because the riffle is expected to be dry with no flow) and need not be specified.

This procedure is repeated until the required number of cross-sections is surveyed. If flows change during the survey, stage at one site should be recorded throughout the day so that this can be related to the time and flow of each cross-section survey (see Varying flows later).

7.3.1 Braided/divided channels

In braided rivers, each channel in a braided reach is treated as a separate cross-section, with temporary staff gauges in each channel. Channels are analysed separately with survey flows, rating curves, and stages of zero flow varying at each cross-section.

A divided channel occurs where a river flows around an island. If it is found that the level variation with flow in each channel is similar, the channels can be treated as one continuous cross-section. For water surface profile analysis, it may also be necessary to ‘dogleg’ the cross-section so that the water level in each channel is the same. Diagonal riffles also create a situation that can be treated as a divided channel, where the water level and depth/velocity distribution are different on either side of the riffle. In such a case, the edge of the riffle (usually the downstream edge) is treated as a vertical wall when modelling and each side is treated as a separate cross-section (Fig. 7.10).

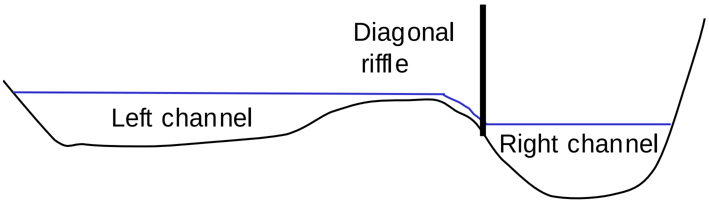


Figure 7.10: **Using divided channels to model a diagonal riffle where the variation of water level with flow is different in each channel.**

7.4 Prediction of water level

7.4.1 Rating curves

After the initial survey, two or more follow up surveys are required to measure water levels so that stage-discharge rating curves can be fitted. Stage-discharge curves are used to predict the variation of water level with flow; 1) at each cross-section, for habitat surveys based either on representative reaches or stratified sampling (habitat mapping), and 2) for the boundary cross-sections for water surface profile modelling or 2D modelling. The method with individual rating curves for each cross-section is known as IFG4 in PHABSIM and is particularly suited to high gradient streams where water surface profile or 2D modelling is difficult. It is also useful to develop rating curves for all cross-sections in a water surface profile analysis because they can be used to determine how roughness varies with flow between each pair of cross-sections.

Stage-discharge calibration field measurements should be done as soon as possible after the main survey to minimise the chance of rating changes occurring between the survey and follow-up measurements. However, there must be a measurable flow and water level change between the follow-up surveys. Water level changes of 50 mm or more are ideal, although smaller changes can be used depending on the size of river and accuracy of flow and level measurements.

On the follow up visits, flow is measured at a good gauging site and the water level at each cross-section (or downstream cross-section for water surface profile analysis) is measured. Bench marks and temporary gauge levels should be checked against the original survey in the field and the source of any discrepancy determined, as this could be either survey error, or movement of the staff gauge or benchmark.

Stage-discharge curves are usually derived by fitting a curve to a series of measurements of water levels and flows. This is routinely done at most river flow recording sites around New Zealand and over time it is possible to develop a curve based on measurements of the full range of flows and water levels experienced at that site. Stage-discharge curves for instream habitat assessments are based on fewer measurements than curves at flow recording sites. This is because:

1. high flow predictions are not usually required
2. results are usually required within a set time
3. measurements must be taken before a flood alters the stage-discharge curve.

The variation of water level with flow over a section of the stage-discharge curve can usually be described by the following relationship:

$\text{Flow} = a(\text{water level - water level at zero flow})^b$

where a and b are constants that give the best fit to the measurements of water level and flow.

This is the best method of deriving the stage-discharge relationship when there are 3 or more measurements taken over the full range of flows for which habitat predictions are to be made.

It is also possible to calculate rating curves that give the best-fit regardless of the estimated water level at zero flow, and curves calculated from the hydraulic properties of the cross-section and variation of Manning's *N* with discharge. Comparison of these curves, and possible adjustments based on professional judgement, result in the best possible estimate of the stage-discharge relationship. Relationships at each cross-section are generally similar and deviations from this 'type curve' will often be the result of data errors.

Stage-discharge relationships usually predict water levels more accurately than water surface profile modelling. There is no restriction on cross-section location, and the process of calibration is simpler than for water surface profile models.

7.4.2 Water surface profile modelling

Water levels at each cross-section along a reach can be calculated using engineering techniques of water surface profile modelling, such as the standard step backwater method (Chow 1959, Henderson 1966). This method is based on the principle of energy conservation. The longitudinal flow profile is calculated from the flow, slope, hydraulic roughness, and the hydraulic properties of the cross-sections. An important assumption in the method is that the distance between cross-sections must be short enough that the hydraulic properties of the cross-sections approximate the hydraulic properties and slope between them, and thus the energy losses. In practice, this means decreasing cross-section spacing at the heads and tails of riffles, where water slopes and cross-section areas change rapidly, and increasing the spacing when the hydraulic conditions are uniform. These conditions are relatively easy to comply with in large low-gradient rivers, but small steep rivers are difficult to model accurately.

The procedures involved in selecting cross-section locations and calibrating the hydraulic model require an understanding of hydraulic principles, and experience. Even then, there are uncertainties in the accuracy of the predictions of the water surface profile, and this limits the range of flows for which predictions can be made.

The hydraulic roughness (Manning's *N*) is determined from field data on discharge, cross-section area, hydraulic radius, and slope. Manning's *N* can vary with flow in an unpredictable manner (e.g., Hicks & Mason 1991) and this limits the range of flows for which the roughness calibration is valid.

The advantage of water surface profile modelling is that all the necessary survey data can be collected on one visit to the site, which is particularly suited to remote locations or rivers where the channel form is often altered by floods and bed movement. There are methods of estimated rating curves at the downstream cross-section, without field measurements at a range of flows. If the downstream cross-section is located at a hydraulic control such as a steep riffle, the rating curve can be estimated with good reliability. Another technique is to measure a number cross-sections below the 'downstream' boundary cross-section, and then use water surface profile modelling to predict the rating curve at the downstream boundary cross-section. In some circumstances, such as upstream of an artificial control on water level that causes a backwater effect, water surface profile modelling is the only practical method.

For water surface profile modelling, cross-sections must describe reach geometry in both longitudinal and cross-sectional profile. This means that a representative reach approach must be used, with the elevation of every cross-section related to the same datum and sections close enough to adequately represent the variation in both the cross-sectional area and longitudinal profile.

If cross-sections are selected with a stratified approach (habitat mapping), the data cannot be used for water surface profile modelling because the longitudinal profile is not defined (i.e., the cross-section water levels have not been measured relative to a common datum and the cross-sections are usually too widely spaced).

7.5 Prediction of water velocities

The distribution of water velocities across a cross-section can be calculated from its conveyance once the water level and flow are known (Fig. 7.11) (Mosley & Jowett 1985). Each velocity can be adjusted for site specific features, such as an upstream obstruction which might cause a reduction in velocity, or a current on a bend increasing local velocities. Each measurement point represents a cell of the total river area (Fig. 5.1), for which the suitability of the velocity, depth, and substrate are evaluated on a scale of 0 (unsuitable) to 1 (optimum). The suitability scores for each of these parameters are then combined (usually by multiplication) to give the point suitability. The point suitability is multiplied by the width of the cell it represents and summed across the cross-section to give the weighted usable width (WUW). The WUW of each cross-section is multiplied by the proportion of the total river length that the cross-section represents, based either on habitat mapping or the distance between cross-sections, to give the cross-section WUA. The total reach WUA is the sum of the WUA of all the cross-sections. Once a hydraulic model of the reach is derived, water velocities and depths can be predicted for any flow and the amount of suitable habitat at that flow evaluated.

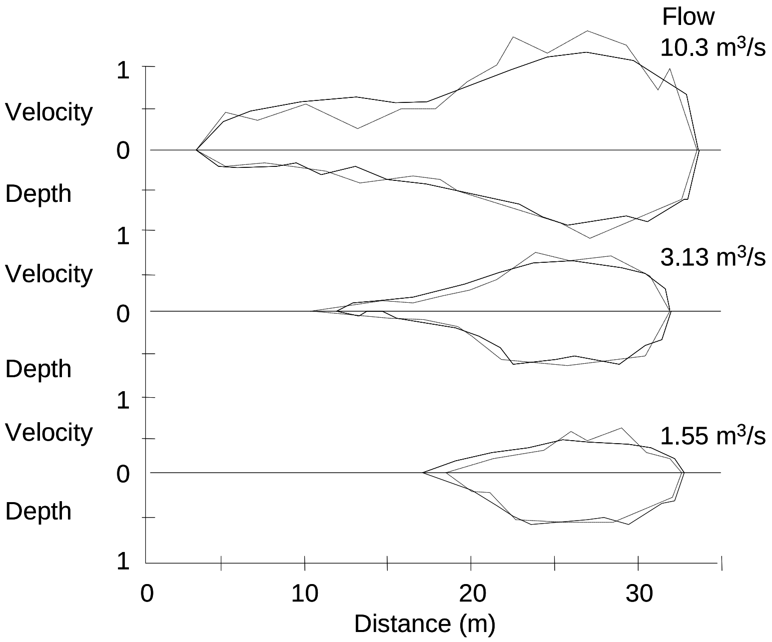


***Figure 7.11*: Comparison of measured velocities and depths (dashed line) with depths and velocities predicted from data measured at three cross-sections at a flow of 13 m^3/s in the Ashley River.**

The computer programme RHYHABSIM can evaluate habitat surveys based on either habitat mapping or representative reaches, with water levels predicted either from stage-discharge relationships or water surface profile modelling. The predictive ability of RHYHABSIM was tested in a braided river. The survey was carried out with flows varying from 26 to 45 m^3/s which reduced the accuracy of predictions (see Section 7.6). When predicted and measured values were compared at 112 point measurements at a flow of 12.8 m^3/s, the average absolute errors in depth and velocity predictions were 0.052 ± 0.046 m and 0.074 ± 0.079 m/s, respectively (Fig. 7.12).

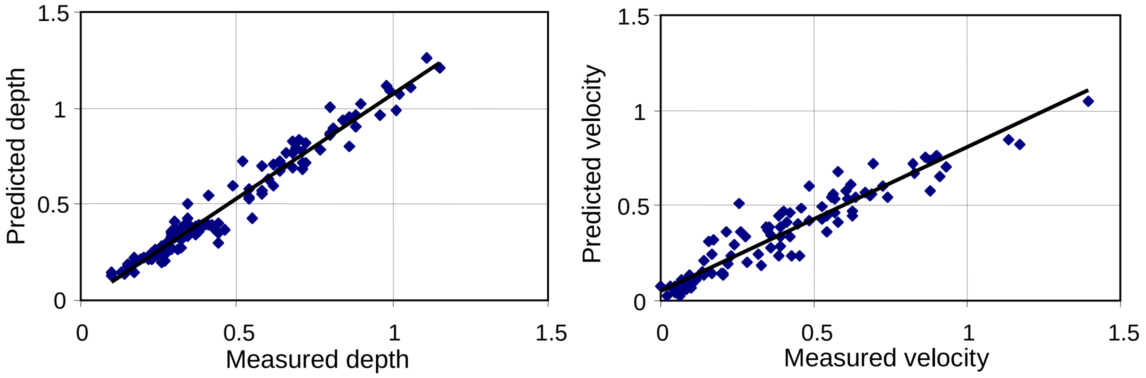


***Figure 7.12*: Comparison of measured and predicted water depths (m) and velocities (m/s) at 12.8 m^3/s in a braided river (Hurunui) predicted from survey data collected over 3 days when the flow varied from 26 m^3/s to 45 m^3/s.**

7.6 Varying flows

Flows can vary spatially within a survey reach, such as when a tributary flow enters the river part way through the reach, or flows can vary during the survey, as they would on a flood recession. In the case where a tributary enters the reach, the calibration and prediction procedures should apply different flows to the sections above and below the tributary input. This type of analysis is easiest using stratified sampling procedures (habitat mapping; IFG4) in 1D programmes. It is difficult using water surface profile and 2D modelling techniques and in these situations the reach would usually be divided into reaches above and below the tributary.

Time varying flows (unsteady) complicate survey procedures because the flow is changing while measuring velocities across a section (usually not by much), but more importantly, because the flow changes between cross-sections, and that makes it difficult to ascertain the actual flow at the time the cross-section, and water level, was measured. For this reason, it is advisable to record the time of all measurements, as well as recording the water level (and flow if possible) continuously through the survey. With a continuous record of flow during the survey, it is possible to calculate the flow at which each cross-section was surveyed and to use this as the ‘survey’ or ‘best estimate’ flow. The procedure is repeated for each cross-section in order to derive the cross-section stage-discharge curve (see Section 7.4.1). Flow must be constant during the collection of water levels for the calibration of water surface profiles, for 1D or 2D models, although flows can vary while surveying bed profiles.

7.7 Fluctuating flow analysis

Fluctuations in flow create a varial zone that is wetted and dried as water levels rise and fall. With frequent flow fluctuations, this zone will not sustain immobile plant and invertebrate species. Mobile species such as fish, and probably some invertebrate species can make some use of this zone, especially for feeding in recently inundated areas of river bed where terrestrial invertebrates in the substrate may have been caught by rising water levels. However, a varial zone that is wetted and dried at more frequent intervals than weekly is expected to usually be unproductive and can be regarded as lost habitat. If it is assumed that biota are immobile, it is possible to calculate the locations within a river that consistently provide suitable physical habitat under fluctuating flows.

The concept is that most aquatic organisms become established at locations that provide suitable habitat at an existing flow. If the flow fluctuates, the velocity and depth at the location occupied by the aquatic organism will change and if that location no longer provides suitable habitat, then that location would not be considered suitable under a fluctuating flow regime.

The numerical evaluation of habitat suitability is to sum the available habitat over a reach, assuming that the habitat value of each location is the minimum habitat value of that location over the range of fluctuating flows.

The calculation of habitat suitability with fluctuating flows involves setting a 'normal' flow (Q_B) and a range around which the flow fluctuates. The habitat at each point (*HSI*) is calculated at Q_B and for flows Q_B ± the fluctuation (e.g., $Q_B – Q_{low}$ and $Q_B + Q_{high}$), where Q_{low} and Q_{high} are the upper and lower fluctuations for a proportion of the fluctuating flow range, Q_V. The *HSI* for the point with fluctuation is $min(Q_B ,\ Q_B + Q_{V,}\ Q_B - Q_V)$. The habitat suitability indices, *HSI*, are multiplied by the area they represent and then summed over the reach to give a single value of WUA for Q_V. When this is repeated for a range of values of Q_V, the results can be plotted as a graph of the effect of flow fluctuations on the amount of habitat at 'normal' flow (Fig. 7.13). The 'normal' flow is the flow that would occur without flow fluctuation and is usually a modal or median flow. The results of a fluctuating habitat analysis are usually expressed in terms of the proportion of habitat at 'normal' flow, where 100% of the habitat is normally available (i.e. if flows were steady), but less is available as flow fluctuations increase.

In the following example (Table 7.1 and Fig. 7.13), the amount of habitat for larval *Aphrophila* (a true fly) was calculated for flows fluctuating from 4 to 8 m^3/s about a normal flow of 5 m^3/s. In Table 7.1, WUA at the normal flow of 5 m^3/s is always greater than or equal to WUA with fluctuation. The amount of habitat with 10% flow fluctuation (i.e., from 4.8 to 5.6 m^3/s) is the minimum habitat value (*HSI*) at flows of 4.8, 5, and 5.6 m^3/s (i.e., 6.60 m^2/m, or 95.35 % of that at the 'normal' flow).

Table 7.1: Variation in weighted usable area (WUA) for *Aphrophila* (a true fly larvae) with flows fluctuating from 4 to 8 m^3/s about a normal flow of 5 m^3/s.

Flow (m^3/s)	WUA without fluctuation (m^2/m)	WUA with fluctuation (m^2/m)	% of WUA at 'normal' flow
4.0	6.75	5.40	78.02
4.2	6.80	5.70	82.31
4.4	6.85	6.00	86.65
4.6	6.88	6.30	90.95
4.8	6.90	6.60	95.35
5.0	6.93	6.93	100.00
5.6	6.94	6.60	95.35
6.2	6.92	6.30	90.95
6.8	6.91	6.00	86.65
7.4	6.86	5.70	82.31
8.0	6.75	5.40	78.02

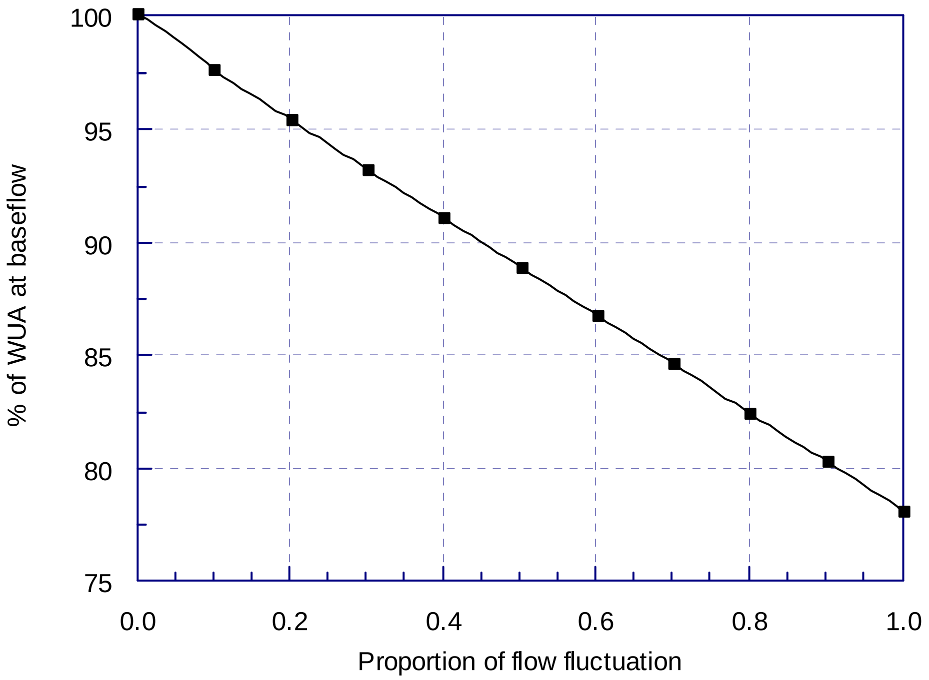


Figure 7.13**: Variation in weighted usable area for *Aphrophila* (a true fly larvae) with flows fluctuating from 4 to 8 m^3/s about a normal flow of 5 m^3/s.**

7.8 Flushing flow analysis

Flushing flows are flows that remove the fine sediments and periphyton accumulations from stream substrates. Flushing flows are necessary in most alluvial streams to remove accumulated fine sediments and to restore interstitial space in gravel substrates. In spring-fed streams, the flow regime is stable and floods and flushing flows are practically non-existent. Suspended sediment concentrations in spring-fed streams are naturally very low so there is no accumulation of sediment that needs flushing. In addition, there is little shallow water with coarse substrate to encourage periphyton growth and spring-fed streams are often dominated by macrophytes.

Surface flushing flows remove the fine sediments from the surface layer, leaving the armour layer largely intact. Periphyton can also be removed by the abrasive action of fine sediments moving over the surface. Channel maintenance flows or deep flushing flows disturb the armour layer, removing the sediments that have deposited within the gravel matrix, as well as maintaining river morphology, particularly the pool/riffle structure.

Flushing flows have both a beneficial and detrimental effect on rivers. In the short-term, they result in a loss of productivity, but in the long-term biota benefit through the improvement in habitat quality. The detrimental effect of high flows on stream biota is largely a result of the high water velocities and bed sediment movement (Jowett & Richardson, 1989; Scrimgeour & Winterbourn 1989). In many New Zealand rivers physical abrasion by sediment movement is probably more important than the effect of velocity alone (Scarsbrook 1995, Jowett & Biggs 1997, Biggs et al. 1999).

Flushing flow or channel maintenance flows cause movement over part of the stream bed only (except in uniform channels with uniform substrate). Sediment transport occurs at practically all flows and as the flow increases the amount and size of sediment transported increases. Some areas of the stream bed will resist movement more than others, so that the area of a stream bed that is disturbed by high flows gradually increases as the flow increases. A suitable flushing flow might be the flow that flushes 80% of the river bed that is submerged at base flow. The area that is to be flushed is an arbitrary decision that must be made when deciding on a flow. Clausen & Plew (2004) calculated the bed-moving or channel maintenance flow (that which moves 84% of the bed sediment) in 41 New Zealand rivers to be about 10 times the mean flow on average, or 40% of the mean annual maximum flow. This can serve as a first estimate of the flow required for maintenance of channel morphology, but individual rivers can differ from this value.

The calculation of the amount of disturbance caused by a flow is based on bed shear stress. Shields showed that particles were likely to move when the dimensionless bed shear stress equalled 0.056. Subsequent studies indicated that this value may be slightly high. Milhous (1998) used data from a small gravel bed stream to show that surface sediments were flushed when the dimensionless bed shear stress exceeded 0.021 and that the armour layer was disturbed when the stress exceeded 0.035. These values are used to calculate the area of the streambed that is flushed by a given flow.

The bed shear stresses are the forces that resist the effect of gravity on water flow. The sum of the bed shear stresses is proportional to the depth of water and the slope of the river as shown in the following equation:

$$\text{dimensionless bed shear stress} = RS/(sg-1)/\text{substrate armour size}$$

where R is the hydraulic radius, S the slope, *substrate armour size* is usually the d85 size, and sg is the specific gravity of the substrate, usually taken as 2.65.

Thus, the slope of the river must be known before flushing flow requirements can be estimated.

RHYHABSIM has two methods of calculating bed shear stress. One method assumes that the average water surface slope is constant over the whole reach, the other estimates local water surface slope using Manning's equation with point velocities and substrate size to estimate roughness, as described below.

Flushing usually occurs at flows higher than the flow at which the instream habitat survey was carried out. The average water surface slope method should be used if the flushing flows are an order of magnitude higher than the survey flow.

Alternatively, the velocity method can be used where the slope is calculated indirectly from velocity and substrate measurements. Bed shear stresses can be calculated from mean column velocity V, Hydraulic radius R, and Manning's N as follows:

$$\text{bed shear stress} = \mathrm{sqrt}(g) * V * \text{Manning's } N / R^{1/6},$$

where g is the acceleration due to gravity.

Manning's N can be calculated in two ways:

1. Strickler's equation

$$N = 0.04145 * d^{1/6}$$

2. Manning's N calculated from the cross-section geometry (mean velocity V and hydraulic radius R) and longitudinal flow profile to give the slope (S):

$$N = 1/V * R^{2/3} * S^{1/2}.$$

The velocity distribution (V at points across the river) at flushing flows should be assumed to be more uniform than that measured at lower flows because velocity variation generally reduces at high flows.

The size of suspended and bedload sediments moved by a flow are calculated from the following formulae presented by Milhous (1998):

$$Max.\ suspended\ sediment\ size = RS/g/((sg-1)\ 0.28)$$

$$Max.\ bedload\ size = substrate\ armour\ size\ (RS/((sg-1)\ 0.018))^{2.85}$$

The term (RS /(sg -1) is equivalent to the dimensionless bed shear stress (see above) multiplied by the substrate armour size.

RHYHABSIM has the option of using an alternative equation (Gessler 1970) to predict sediment movement. This method predicts the probability of movement (or of remaining in place) for a given sediment size and shear stress.

This implementation incorporates a hiding factor, under the assumption that large substrates 'hide' small substrate from the effects of the current. There is good agreement between the Gessler (1970) and Milhous (1998) methods, with Gessler's method having the advantage that it predicts the probability of movement for all sediment sizes. The hiding factor is incorporated into the calculations in the term:

$(d_i/d_{50})^h$

where d_i is substrate size and h is the hiding factor..

The hiding factor increases the effective shear stress on small particles to allow for the hiding effect of the larger particles. Values of the exponent h could vary from 0.113 (Andrews 1984) and 0.33 (Duncan & Biggs 1998). A value of 0.113 is used in RHYHABSIM.

7.9 Confidence limits

Confidence limits can be placed on instream habitat predictions. Estimates of confidence limits are based on the assumption that cross-section locations are selected randomly and the bootstrapping method selects random combinations of cross-sections to calculate instream habitat and thus variability. These statistical confidence limits reflect the variability in cross-section properties and do not address all uncertainties in instream habitat modelling. For example, habitat suitability criteria have more influence on the shape of the WUA/flow curve than the number or selection of cross-sections (provided the guidelines in Section 7.2 are followed) and the uncertainty in these criteria depends on the amount and quality of the data that they are based on.

In the randomisation process, cross-sections are selected with replacement. This means that a bootstrapped sample could, by chance, replace all the (different) cross-sections with the same cross-section – an unlikely scenario by valid under the bootstrap method. If the river is comprised of pools, riffles and runs and cross-sections were randomly selected from all the cross-sections in the reach, the assumption of random selection of cross-sections would be invalid because the cross-sections within the reach were not randomly chosen – they were selected to represent pools, runs and riffles. However, it is possible in bootstrapping to randomly select cross-sections within each of the mesohabitat types and this is the procedure used in RHYHABSIM.

With stratified random sampling the mean value is calculated as the weighted average over all mesohabitat types:

$$\bar{x} = \sum_{i=1}^{m} w_i x_i$$

Where $\bar{x}$ is the overall reach mean, w_i the weight applied to mesohabitat type i, and x_i is the mean of cross-section values in mesohabitat type i in a reach of m mesohabitat types. The weight w_i is the proportion of river reach length represented by that mesohabitat type, so that the sum of the weights over the reach equals 1. Individual cross-section weights within each mesohabitat type are equal and their sum equals w_i.

The standard error within each mesohabitat type is the square root of the sample variance (i.e. standard deviation) divided by the square root of the number of cross-sections in the mesohabitat type n_i:

$$s_i = \frac{\sqrt{\dfrac{\sum_{j=1}^{n_i} (x_j - x_i)^2}{n_i - 1}}}{\sqrt{n_i}}$$

The overall standard error s of the reach mean $\bar{x}$ is:

$$s = \sqrt{\sum_{i=1}^{m} \left(\frac{w_i}{w} \right)^2 s_i^2}$$

Where s_i is the standard error of the mean within the i^{th} of m habitat strata, and w is the sum of weights in the reach (usually 1).

Confidence limits for the overall mean are:

$$CL_{mean} = \bar{x} \pm s^2 t_{[\alpha, n-1]}$$

Where t is the t-statistic for the whole sample (n cross-sections) calculated by the bootstrap-t method described by Manly (1997) and adapted for stratified samples.

These confidence limits indicate the confidence that can be placed on the predicted value (of WUA,) at a particular flow, assuming that cross-sections have been randomly selected within each stratum. In practice, selection within a stratum tries to encompass the range of variation within the stratum thus reducing the uncertainty that would be associated with truly random sampling.

In evaluation of flow requirements, the shape of the habitat/flow curve is of more interest than the actual amount of habitat, and as shown in Section 7.2 fewer cross-sections are needed to define the shape than are needed to stabilise the magnitude of WUA predictions.

Confidence limits on the shape of the habitat/flow curve can be estimated by bootstrap methods applied to the slope the curve. For example, maximum habitat is at the point where the slope is zero. The procedure is to calculate confidence limits for the slope at each point along the habitat/flow relationship. The upper and lower confidence limits on slope are then used to estimate the range of flows that bound the calculated slope. Figure 7.13 shows the calculated relationship between flow and slope and the upper and lower bounds on that relationship. The points Q_L and Q_U indicate the confidence limits on the slope at the flow Q_1. If Q_1 has a slope of zero (the maxima), then we can be confident that the habitat maxima lies between Q_L and Q_U.

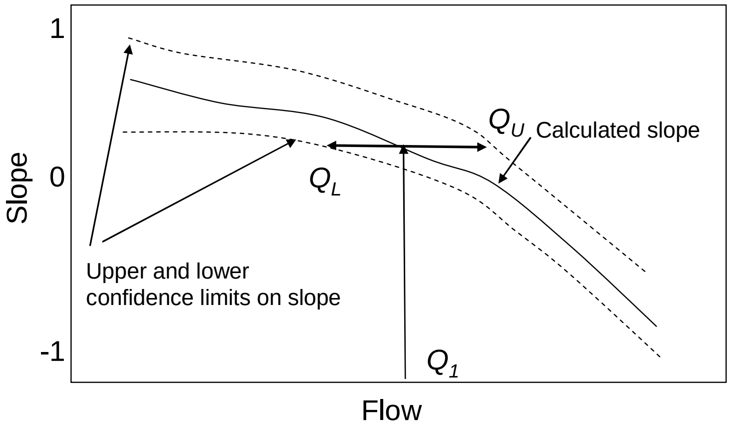


***Figure 7.14*:** **Relationships between slope of habitat/flow relationships and flow and an example of determining the confidence limits on flows.**

8. Generalised instream habitat models

Within the suite of instream habitat models, it is possible to select the model that is appropriate to the situation. In many cases, the simple generalised model, with one measurement of width and flow, can be used to define a minimum flow for the appropriate critical values and habitat retention levels. If the stream morphology is unusual (i.e., substantially different from the range of rivers used to derive the generalised model) or if greater certainty is required, the width can be measured at two flows and WAIORA used to apply the generalised models. Finally, if the value of the instream or out-of-stream resource requires the most detailed level of consideration, instream habitat surveys and 1D, or even 2D, models can be used to predict habitat response curves or net rate of energy intake in fish models (Hayes et al. 2003, Hayes et al. 2007).

Conventional instream habitat models link hydraulic models to habitat suitability curves for water depth, velocity and bed particle size. The hydraulic model predicts the values of point habitat variables (velocity, depth, particle size) for a given discharge in a stream reach. Suitability curves are used to calculate point habitat values for each combination of point habitat variables. Their product is a habitat suitability index (HSI, ranging between 0 and 1), and when summed over the reach surface area, HSI gives the weighted usable area (WUA) which can be simulated over a range of flows to give reach-scale relationships between WUA and discharge. Section 5 gives a more detailed description of hydraulic habitat modelling.

Applying conventional instream models in a stream reach requires considerable field effort and experience. At the least, they require the measurement of water depth and velocity across a number of cross-sections, plus calibration measurements, for which cross-section water levels need to be measured at 2 or more flows. However, data requirements can be even more onerous, e.g., 2D hydraulic models generally require extensive surveys of bed topography plus calibration water level measurements. Several approaches have been proposed for reducing this effort. Some are based on a simplification of the hydraulic complexity within the reach, by using hydraulic geometry relationships and considering point velocities as equal to their average (Jowett 1998), or simplifying their statistical distribution (Singh & Broeren 1989; Lamouroux et al. 1998). Others try to identify general patterns in existing applications of the models (Hatfield & Bruce 2000). Lamouroux & Capra (2002) proposed to model directly the output of a conventional instream habitat model using simplified and cost-effective reach descriptions (depth- and width-discharge relationships, particle size, median flow). The advantage of the resulting generalised habitat models is that no simplifying hypothesis is made on the distribution of hydraulic variables within reaches. Their use requires little experience or field effort, and the models provide HSI and WUA curves which can be interpreted in a similar way as conventional ones.

Tests of generalised models in France (Lamouroux & Capra 2002) and New Zealand (Lamouroux & Jowett 2005) found that habitat values for taxa were predictable from simplified hydraulic data. Reach hydraulic geometry (mean depth and mean width-discharge relationships), average bed particle size and mean flow could be used to provide reliable estimates of habitat values in natural stream reaches. Key physical variables driving habitat values were found to be similar in New Zealand and in France. The Reynolds number of reaches (discharge per unit width) governs changes in habitat value within-reaches. The Froude number at mean flow, which indicates the proportion of riffles in stream reaches, was generally the major variable governing overall habitat value in the different reaches. This is consistent with the preference of the benthic fauna, such as many of the native New Zealand fish species and benthic invertebrates, for riffles (Jowett & Richardson 1996; Jowett 2000), and the non-benthic aquatic fauna for runs or pools (e.g., Jowett 2002).

The generalised habitat models were robust. Tests of the French models of Lamouroux & Capra (2002) in New Zealand rivers were very satisfactory, and most New Zealand models gave reasonable accuracy when applied in rivers larger or smaller than those used to calibrate them (with some loss of accuracy for some taxa).

Generalised models necessarily lose some information compared to conventional models such as RHYHABSIM (Fig. 8.1). This loss must be balanced against requirements for field work and experience in conventional modelling. In particular, hydraulic geometry relationships in reaches (as used by generalised models in WAIORA) can be easily obtained from field measurements made at two different discharges or using regional models (Leopold et al. 1964; Jowett 1998; Lamouroux et al. 1998). By combining generalised models and hydraulic geometry relationships, estimating habitat values in multiple streams is possible from few field measurements; detailed topographies of stream reaches, associated velocity measurements and hydraulic model calibration are not required.

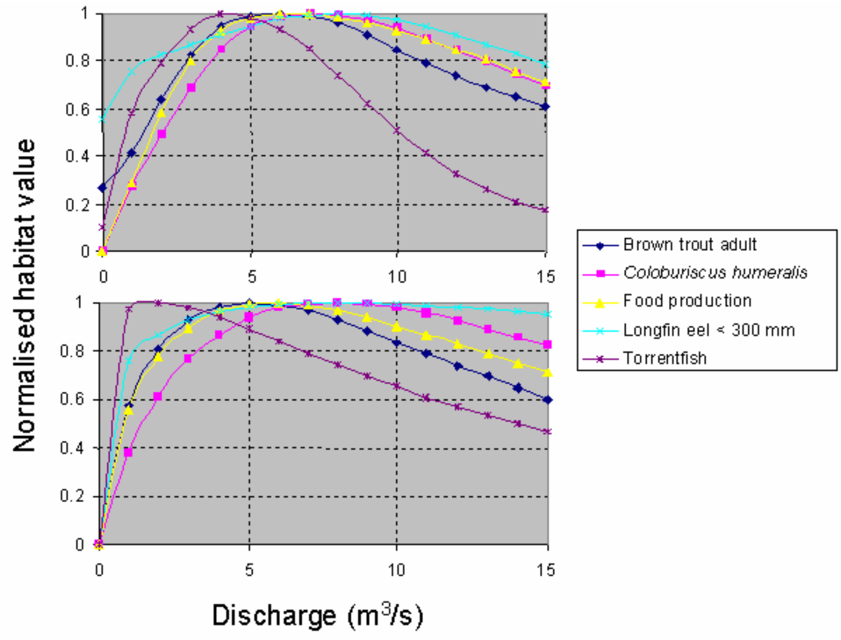


***Figure 8.1*:** **Comparison of normalised habitat per unit width predicted by habitat modelling in RHYHABSIM (upper) and the generalised method (lower).**

Generalised habitat models suggest general, simple rules can be used to improve flow management, or to estimate regulation impacts, over whole river networks. An example of such a rule is that a discharge value of about Q = 0.3*Width would provide optimal habitat values for several freshwater taxa in New Zealand.

8.1 Derivation of generalised habitat model coefficients

The generalised model takes the form:

$$HSI = a \times \left(\frac{Q}{W} \right)^{c} \times e^{-k\frac{Q}{W}}$$

The values *c* and *k* describe the shape of the curve, whereas the parameter *a* is a scaling factor that varies from reach to reach. The values *c* and *k* are of most interest, because the assessment of flow requirements is based on the shape of the curve, rather than the absolute values. The equation has a maximum at *c*/*k*, so that this ratio specifies the discharge per unit width that provides maximum habitat.

The values of model coefficients for each taxa were derived from a dataset of 99 reaches of New Zealand rivers. The reaches in this dataset have mean flows varying from 0.6 m^3/s to 53.8 m^3/s (the same data were used by Lamouroux & Jowett (2005)). Lamouroux & Jowett fitted a non-linear mixed effects model to these data for habitat and flows ranging from 0.05 times the mean flow to the mean flow. This model described a common shape for each taxa (i.e., *c* and *k* were held constant, but *a* was allowed to vary between reaches).

For some taxa, generalised curves could not be developed by the method used in Lamouroux & Jowett (2005) because the flow range that was modelled was too high and did not include the flow that provided maximum habitat. For example, often it was difficult to fit a model for inanga, or other species with low velocity habitat preferences, to survey data from a large river, because the flow in a large river would need to be very low to maximise the area of river with suitably low velocities. An alternative method of deriving generalised curves was used. Instead of fitting one value of *c* and *k* to all reaches, values of *c* and *k* were fitted to each reach. Values for *c* and *k* were then examined and reaches with negative values and outlying values of *c*/*k* were excluded. The median values of *c* and *k* are shown in Table 8.1.

The optimum flow ranges for each taxon in Table 8.1 were calculated using average New Zealand at-a-station hydraulic geometry relationships from Jowett (1998), to give some idea of the size of river that was likely to provide optimum habitat for each of taxon. The calculations required are described below:

Width = $15.8 \pm 7.5 \times \text{Flow}^{0.176 \pm 0.066}$ (from 73 rivers, in Jowett 1998)

Using the upper and lower bounds of the ± standard deviations on the curve parameters to calculate the likely minimum and maximum size of river that would provide maximum habitat. The lower bound flow per unit width is:

Flow per unit width = $\text{Flow} / ((15.8-7.5) \times \text{Flow}^{(0.176-0.066)})$
= $\text{Flow}^{0.89} / 8.3$

The lower bound of flow range is where flow per unit width equals c/k:
$\text{Flow}^{0.89} / 8.3$ = c/k, and
Lower bound of flow range = $(8.3 \times c/k)^{1.1236}$

Similarly, the upper bound to the flow range can be calculated:

Flow per unit width = $\text{Flow} / ((15.8+7.5) \times \text{Flow}^{(0.176+0.066)})$
= $\text{Flow}^{0.76} / 23.3$

The upper bound of flow range is where flow per unit width equals c/k :
$\text{Flow}^{0.76} / 23.3$ = c/k, and
Upper bound of flow range = $(23.3 \times c/k)^{1.316}$

The origins of the habitat suitability curves used to fit the generalised curves are shown in Table 8.2.

Table 8.1: **Generalised habitat models used to predict habitat values (HSI) from average hydraulic geometry characteristics of stream reaches. Model parameters *c* and *k* are developed for each reach independently and the median value selected, excluding reaches with negative values of *c* and *k* and outlying values of *c*/*k*.**

Species	***C***	***k***	**Optimum discharge per unit width**	**Optimum flow range (m^3/s)**	
			(m^2/s)	**Min.**	**Max.**
Inanga	0.19	19.74	0.01	0.06	0.14
Shortjaw kokopu[+]	0.19	16.35	0.01	0.07	0.18
Upland bully	0.11	8.63	0.01	0.08	0.21
Crans bully	0.09	6.84	0.01	0.09	0.22
Banded kokopu (juvenile)	0.19	13.3	0.01	0.09	0.23
*Galaxias vulgari*s	0.03	2.29	0.01	0.09	0.25
Roundhead galaxias	0.31	10.64	0.03	0.21	0.61
Flathead galaxias	0.28	9.11	0.03	0.21	0.64
Longfin eel (< 30cm)	0.07	2.07	0.03	0.24	0.72
Lowland longjaw galaxias	0.33	9.35	0.04	0.25	0.77
Redfin bully	0.26	7.39	0.04	0.25	0.77
Shortfin eel (< 30cm)	0.13	2.32	0.05	0.41	1.37
Common bully	0.39	6.51	0.06	0.46	1.55
Brown trout fry	0.86	10.21	0.08	0.67	2.42
Brown trout yearling	0.40	4.18	0.09	0.76	2.82
Nesameletus[*]	0.26	2.62	0.10	0.80	2.98
Brown trout spawning	1.24	9.89	0.13	1.05	4.11
Bluegill bully	1.01	6.13	0.16	1.42	5.88
Rainbow trout spawning	1.49	8.78	0.17	1.47	6.12
Deleatidium[*]	0.33	1.92	0.17	1.50	6.25
Torrentfish	0.88	4.05	0.22	1.95	8.49
Brown trout adult	1.17	4.35	0.27	2.46	11.18
Food producing habitat	1.19	4.25	0.28	2.57	11.77
Rainbow trout feeding (30–40 cm)	0.93	2.89	0.32	3.02	14.19
Coloburiscus humeralis[*]	1.35	4.17	0.32	3.02	14.22
Aoteapsyche[*]	1.44	3.17	0.45	4.44	22.29
Zelandoperla[*]	1.71	3.40	0.50	4.97	25.43

[*] large river habitat suitability curves (see Jowett 2000).

[+] suitability for cover locations only

Table 8.2: **Source of data for habitat suitability criteria used for the development of generalised habitat curves.**

Species	**Reference**
Aoteapsyche	Jowett et al. 1991
Banded kopopu (juvenile)	McCullough 1998
Bluegill bully	Jowett & Richardson 1995
Brown trout adult	Hayes & Jowett 1994
Brown trout fry	Raleigh et al. 1986
Brown trout spawning	Shirvell & Dungey 1983
Brown trout yearling	Raleigh et al. 1986
Coloburiscus humeralis	Jowett et al. 1991
Common bully	Jowett & Richardson 1995
Crans bully	Jowett & Richardson 1995
Deleatidium	Jowett et al. 1991
Flathead galaxias	Baker et al. 2003
Food producing habitat	Waters 1976
*Galaxias vulgari*s	Jowett & Richardson 1995
Inanga	Jowett 2002
Longfin eel (< 30cm)	Jowett & Richardson 1995
Lowland longjaw galaxias	Baker et al. 2003
Nesameletus	Jowett et al. 1991
Rainbow trout feeding (30–40 cm)	Thomas & Bovee 1993
Rainbow trout spawning	Jowett et al. 1996b
Redfin bully	Jowett & Richardson 1995
Roundhead galaxias	Baker et al. 2003
Shortfin eel (< 30cm)	Jowett & Richardson 1995
Shortjaw kokopu	McDowall et al. 1996
Torrentfish	Jowett & Richardson 1995
Upland bully	Jowett & Richardson 1995
Zelandoperla	Jowett et al. 1991

9. Comparison of hydraulic modelling methods

Key points about 1D and 2D hydraulic models

- If done well, 1D and 2D surveys produce similar results.
- For a 1D survey, stratified random sampling and development of rating curves at each cross-section is a robust method.
- Complex hydraulic methods such as water surface profile modelling and 2D modelling require high skill levels in surveying the appropriate features in bed topography and hydraulic calibration procedures are subjective.
- Although the spatial resolution of 2D models is usually greater than that of 1D models, there is no reason why the spatial resolution of a 1D model could not equal that of a 2D model.
- High spatial resolution may not be necessary because studies show that relatively few cross-sections produce robust predictions.
- Potentially, 2D models should be able to predict complex changes in flow patterns and extrapolate to higher flows than a 1D model.
- 2D modelling applies to a representative reach, whereas a 1D survey by habitat mapping can apply to a larger segment of river.

Hydraulic modelling is used to predict water depths and velocities in a reach or segment of river over a range of flows. These predictions are then used to show how usable habitat varies with flow. In order of increasing cost and complexity, hydraulic models range from simple relationships, such as hydraulic geometry, that predict average depth and velocity to 1D models, 2D models and even 3D models. Regardless of the quality and complexity of the hydraulic models, their utility is limited by the current state of biological knowledge and models that link population response to hydraulic change.

9.1 Hydraulic geometry

Relationships between flow and average depth and velocity are probably the simplest means of assessing the way hydraulic habitat changes with flow. This method requires relatively few measurements compared to IFIM survey techniques. In two trials of this method, mean depths and velocities were within 15% of the values predicted by IFIM surveys over a range of flows from half to twice the calibration flows (Fig. 9.1). Milhous et al. (1989) compared predictions of habitat suitability (weighted usable area) calculated using hydraulic geometry relationships with habitat suitability calculated from IFIM surveys and found they predicted similar patterns of variation of habitat with flow, with habitat values that were usually within 20% of each other.

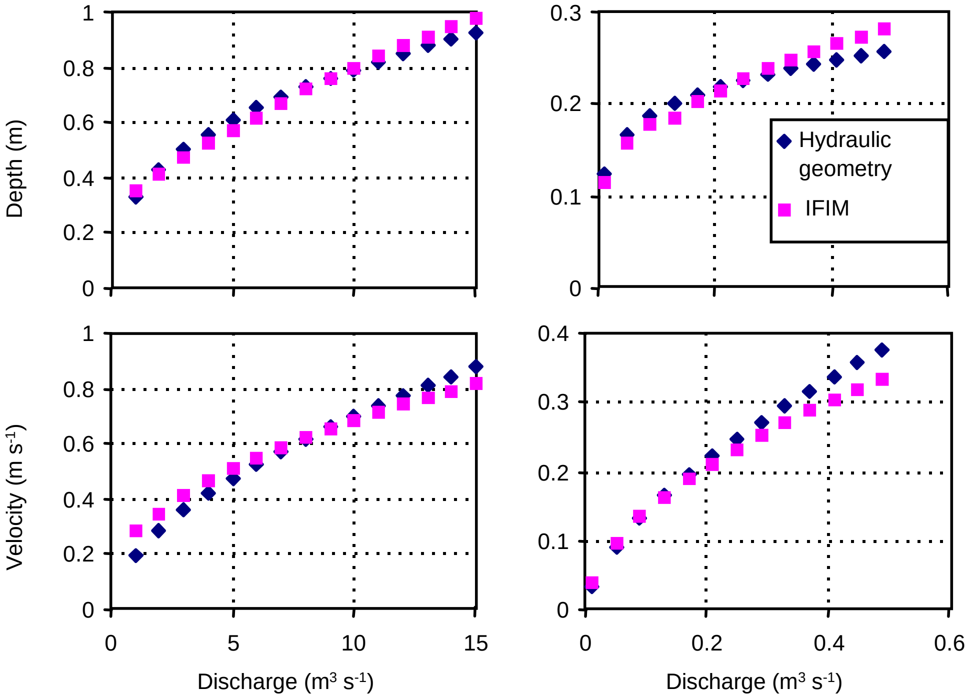


Figure 9.1: **Mean depth and velocity in the Kuratau River (left) and Cosseys Creek (right) predicted by RHYHABSIM (1D model) based on an instream habitat survey, compared with mean depth and velocity predicted by hydraulic geometry relationships estimated by the 'quick' survey method, at flows of 5.32 and 10 m^3/s in the Kuratau River and flows of 0.122 m^3/s and 0.249 m^3/s in Cosseys Creek.**

Hydraulic geometry provides a method for making an initial assessment of environmental impact of proposed flow changes, provided habitat requirements can be specified in terms of mean velocity and depth. Hydraulic geometry essentially quantifies macro-habitat. While many fish and stream insects undoubtedly make use of habitat on a micro scale, many of the features that create microhabitat, such as substrate, bed, and bank forms, vary little with flow and a flow requirement that provides suitable macro-habitat should also provide suitable microhabitat.

Calculation of velocity and depth from at-a-station hydraulic relationships (Section 3.3) assumes that the power law relationships between stream width and average depth and between discharge and average depth hold over the required range of flows. The relationships will be inaccurate where there are changes in cross-section geometry (e.g., a significant change in bank slope) that are outside the range of calibration flows.

Habitat suitability criteria specify a range of suitable water depths and velocities (e.g. Fig. 6.4) and these can be used to give a flow (or range of flows) that provides maximum weighted usable area. Habitat suitability criteria can be used to define threshold levels of depth and velocity, below which habitat quality begins to decline. A simple method of flow assessment based on hydraulic geometry can be used as a preliminary means of indicating whether average hydraulic conditions, resulting from a change in flow, are 'safe' or approaching a 'threshold', such as a minimum acceptable depth or velocity, thus predicating the need for more extensive habitat survey and analysis.

9.2 1D and 2D hydraulic habitat models

As discussed in the sections above, several computer models have been developed for the evaluation of physical habitat, water temperature, and sediment processes. Current 1D model software includes PHABSIM (physical habitat simulation; Bovee 1982; Milhous et al. 1989) and RHABSIM (river habitat simulation) used in the United States, RHYHABSIM (river hydraulic habitat simulation; Jowett 1989, Clausen et al. 2004) used in New Zealand, EVHA (Evaluation of Habitat; Ginot 1998) in France, CASIMIR in Germany (habitat calculation only) (Jorde 1997), and RSS (River simulation system; Killingtviet & Harby 1994) in Norway. 2D and 3D modelling software is also now available (e.g., River2D (Steffler et al. 2003: www.river2d.ualberta.ca), and Hydro2de – NIWA's 2D model (Beffa 1996; Duncan & Carter 1997); SSIIM (3D model: www.bygg.ntnu.no/~nilsol/ssiimwin). More recently, individual-based fish models (Railsback and Dixon 2003) and models based on energetic concepts (Addley 1993; Guensch et al. 2001; Hayes et al. 2000, 2003, 2007) have been developed to the stage where they could be used for flow assessment.

In braided rivers, a 2D model has the advantage of being able to predict braiding patterns and the proportion of flow in each of the braids, whereas a 1D model is limited to the range of flows that are contained within the surveyed channels. Although 2D models usually predict water velocities reasonably accurately as shown in Fig. 9.2, Williams (2001) pointed out that velocity prediction was poor (r^2=0.09) in a 2D model of a 1500 m reach of shallow pools and riffles that was developed by Guay et al. (2000). Guay et al. (2001) later attributed inaccuracy to highly turbulent currents, shallow water, complex riverbanks, and a riverbed of highly variable roughness on a small spatial scale. Tarbet & Hardy (1996) developed a 2D model of the Logan River and compared measured and predicted depths and velocities at 136 points at a flow of 7.7 m^3/s and 150 points at a flow of 4.2 m^3/s. They found that at 4.2 m^3/s, the modal error in velocity was 0.6 m/s with a modal depth error of 0.25 m, and at 7.7 m^3/s the velocity error was 0.15 m/s and depth error 1 m.

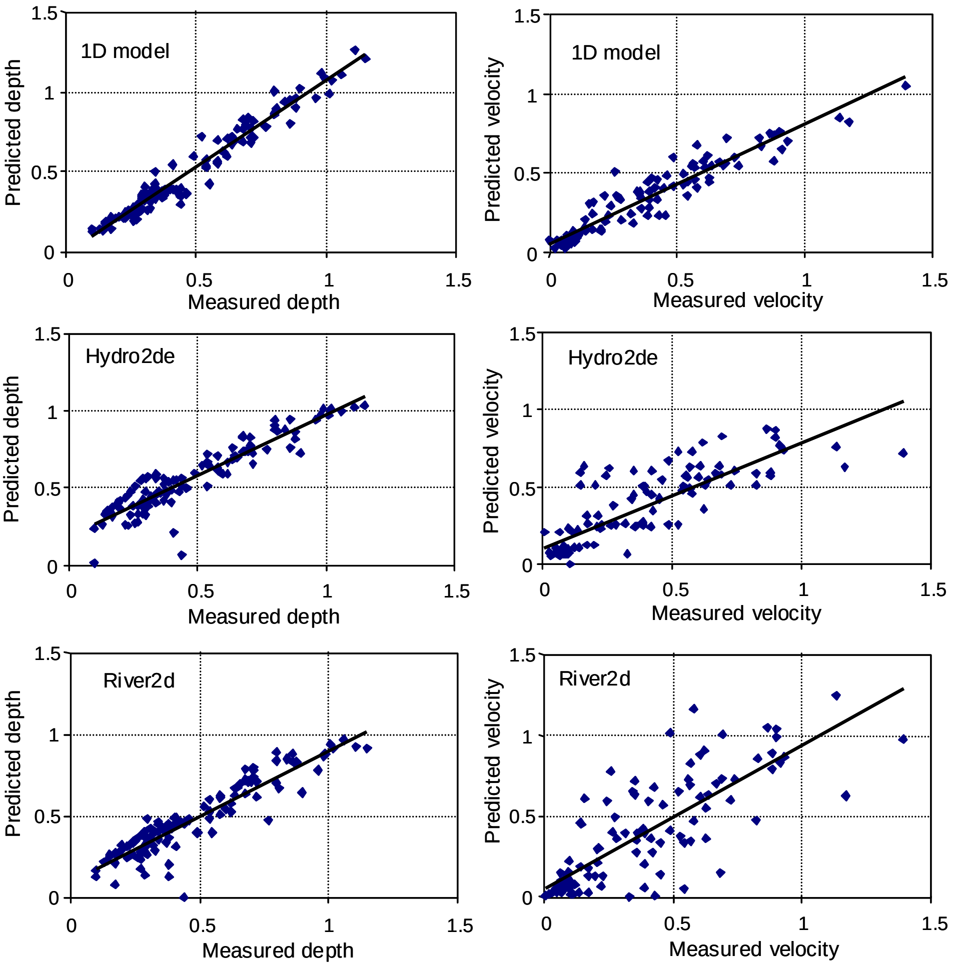


***Figure 9.2*:** **Comparison of measured and predicted water depths (m) and velocities (m s^{-1}) at 12.8 m^3 s^{-1} in a braided section of the Hurunui River using a 1D model RHYHABSIM, Hydro2de, and River2D.**

In any modelling, the quality of the results will depend on the quality of the field work and calibration. This is especially true of 2D models where, the accuracy of the topographic model has a major effect on the accuracy of depth and velocity predictions. In gravel bed rivers, the accuracy of velocity prediction using 1D and 2D models is similar. In the Ashley River, Mosley and Jowett (1985) predicted depths within ±0.03 m and velocities with an average absolute error of about ±0.15 m/s at flows ranging from 0.083 m^3/s to 14.4 m^3/s with a 1D model (RHYHABSIM). In the Rangitata River, a 2D model predicted depths and velocities with average absolute errors of 0.063 m and 0.18 m/s, respectively. In a 1D model, replication of measured water depths and velocities is exact when the measured flow is simulated (with RHYHABSIM). In a 2D model, it is difficult to calibrate the model so that measured water surface levels are modelled precisely, and any error in water surface level translates to an error in predicted depth and mean cross-section velocity. 1D models are easier to calibrate and predict water surface level more accurately than 2D models, at least within the range of rating curve calibration. Within a reach, a 2D model requires more data points than a 1D model and therefore gives a better measure of the longitudinal variations in depth and velocity. As predicted flows depart from the flow used to calibrate a 1D model, uncertainty in velocity distribution increases because it can change with flow. 2D models are likely to predict such changes in velocity distribution more accurately than 1D models, although in both cases, predicted depths and velocities will be incorrect if water surface levels are not modelled accurately.

If done well, 1D and 2D modelling produce similar results (Waddle et al. 2000). In the braided Hurunui River, predictions using RHYHABSIM and Hydro2de were similar, but differed from those using River2D (Figs 9.2 & 9.3). The different results from the two 2D models were probably the result of the calibration process and grid sizes, rather than the models themselves, and this highlights the degree of subjectivity in this form of hydraulic modelling. A comparison of WUA predictions using 1D RHYHABSIM and River2D showed almost no difference in a short reach of the Travers River (Fig. 9.4).

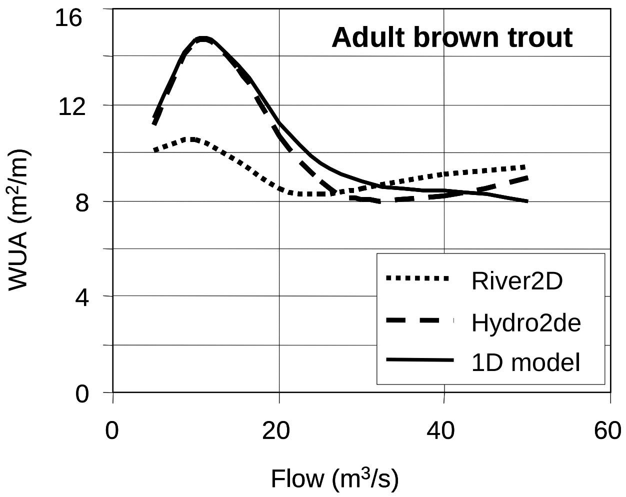


Figure 9.3: **Comparison of adult brown trout habitat (WUA m^2/m) in the Hurunui River predicted by a 1D model (RHYHABSIM) and two 2D models (Hydro2de and River2D); using Hayes & Jowett (1994) adult brown trout habitat suitability criteria.**

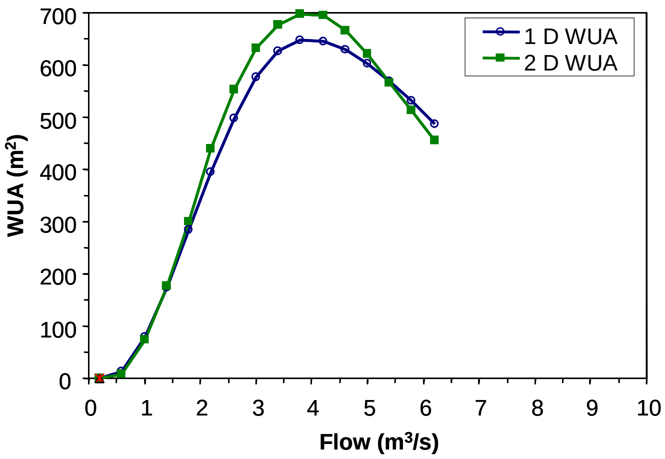


Figure 9.4: **Comparison of adult brown trout habitat (WUA m^2) predicted in 80 m of the Travers River using a 1D (RHYHABSIM) and 2D model (River2D); using Hayes & Jowett (1994) adult brown trout habitat suitability criteria.**

Potentially 2D models should be able to predict complex changes in flow patterns and extrapolate to higher flows than a 1D model. 2D modelling can only be applied to a reach, the length of which is usually up to 1 km; a constraint imposed by survey costs. 1D surveys can be carried out over longer sections of river using the habitat mapping method, so that they can include a greater variety of hydraulic conditions, although these are usually not surveyed to the same level of detail as in a 2D survey.

9.2.1 Spatial definition and integration

In both 1D and 2D models, the basic unit of area is a cell representing a portion of the river with reasonably uniform hydraulic characteristics. In a natural river, the variation in depth and velocity is usually greater laterally than longitudinally, so that an efficient representation of a river will have a greater resolution (i.e., spacing of measurement points) laterally than longitudinally. In a 1D model, the cells are elongated longitudinally, consistent with an efficient representation of the variation in depth and velocity in a river. In 2D models, the cells (usually rectangular or triangular) have similar transverse and longitudinal dimensions for computational reasons (although triangles in triangular irregular mesh models can vary in shape and size).

The method of evaluating habitat suitability and integrating weighted usable area in hydraulic models can vary from model to model depending upon the cell structure and model assumptions. In general, the process is to calculate habitat suitability for the cell and then multiply by the area that the cell represents:

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

Where n is the number of cells, S_i is the composite habitat suitability for the cell i, and A_i is the area represented by the cell.

This calculation appears quite straight-forward, but the spacing of measurements and cell size can influence results. A conventional (PHABSIM) hydraulic analysis calculates the average hydraulic conditions in each cell, evaluates habitat suitability and then sums the product of cell area and habitat suitability over the reach. If the cell size is small, there will be little variation of depth and velocity within the cell and the calculation will be accurate. However, if the cell size is large then the average conditions may not adequately represent the habitat that is actually within the cell. For example, if the cell width is 1 m and the depth varies from 0.02 m to 0.8 m between one side and the other, the average depth will be 0.41 m. The available habitat in this cell for a shallow water species (e.g., depth < 0.3 m) is zero, even though about 40% of the cell provides habitat that is less than 0.3 m in depth.

In River2D (a triangular irregular mesh model), habitat suitability (*csi*) is calculated for each computational node and this suitability is applied to the Thiessen polygon formed with surrounding nodes. In Hydro2de (a rectangular grid model), the cell average hydraulic characteristics are calculated before evaluating habitat suitability and multiplying by cell area. These methods are also susceptible to the averaging problem described above, if individual cells span a considerable variation in depth or velocity.

In 1 D models, the basic calculation of hydraulic and habitat variables assumes that the measured point values represent a larger area - a cell. Different methods of integrating values within a cross-section are available. These are analogous to mid-point and trapezoidal rules of numerical integration. The simplest method is where the cell is the area between two measurement points (left Fig. 9.5).

If the cell (Length = *L*) is between measurement points, the cell area *A* and average velocity *V* is:

$$A = \frac{Y1+Y2}{2} \times (X2 - X1) \times L$$

$$V = \frac{V1+V2}{2}$$

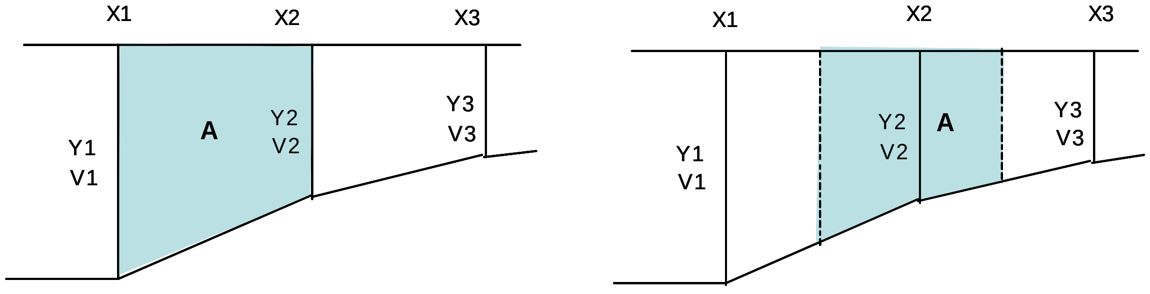
Alternatively, more accurate integration is achieved where each point in a cross-section represents a cell with its width equal to half the distance to adjacent points (right in Fig. 9.5), and the cell area *A* and average velocity *V* is:

$$A = \frac{\left((X2 - X1) \times \left(\frac{Y1+3Y2}{2} \right) + (X3 - X2) \times \left(\frac{3Y2+Y3}{2} \right) \right)}{4} \times L$$

$$V = \frac{{}^{V1}\!/_{2} + V2 + {}^{V3}\!/_{2}}{2}$$

The flow Q_c per cell is:

$$Q_c = \frac{(Y1+Y2) \times (X2 - X1) \times (V1+V2)}{4}$$



Between measurement points

At measurement point

***Figure 9.5*:** **Two possible definitions of cell characteristics in 1D models, where X denotes offset, A area, Y depth, and V velocity.**

PHABSIM has recently (1999) adopted the default assumption that a cell is the area between points and that its depth, velocity etc. is the average of those measured at the cell edges. This tends to smooth variations in depth and velocity.

Cell values, either as points or averages, are an approximation and the degree of potential error will depend on the survey spacing and the habitat suitability curves. To overcome this, RHYHABSIM interpolates values of depth, velocity, and substrate between measured points and integrates habitat suitability over the cell. Water depth, velocity, habitat suitability etc. are interpolated at 10 equally spaced intervals between measurement points. This gives the best possible measure of habitat suitability, assuming linear interpolation is appropriate.

PHABSIM and RHYHABSIM sum cell width weighted values times the section length (or weight) over the reach and then divide by the total reach length as follows:

$$WUA = \frac{\sum_{j=1}^{N}\sum_{i=1}^{n_j} s_i w_i L_j}{\sum_{j=1}^{N} L_j}$$

Where s_i is the cell value, w_i is the cell width for n_i cells for the jth cross-section of N cross-sections in the reach. L_j is the weight (representative length) of the jth cross-section.

Spatial integration methods do not usually have a significant affect on predictions, provided that there are not large differences in water depth and velocity between measurement points.

10. Interpretation of habitat analyses

Key points about instream habitat interpretation

- Instream habitat methods predict depth and velocity and evaluate how well specific depth and velocity suitability criteria are met by different flows.
- The criteria may describe habitat for the target species/life stage with the added aim of providing for organisms with lower flow requirements, or can be regarded as representing general instream conditions that are considered appropriate for the ecological function.
- Interpretation of results can be difficult. Do not be too prescriptive or literal in identifying maximum habitat area.
- Consider the effect of flows on organisms, their habitat and food supply. Consider only flow-related requirements.
- Remember that the habitat suitability criteria are the most influential step in the process.

The procedures involved in carrying out a habitat survey, deriving habitat suitability curves, modelling instream habitat for a range of flows, and calculating the variation of habitat with flow are relatively straight forward, especially with the computer programmes now available. Interpretation of the habitat-flow relationships and the assessment of an appropriate minimum flow for a river is far from straight forward and unfortunately there are no computer programmes that assist with this step.

10.1 Weighted usable area

Weighted usable area (WUA) has been an unfortunate choice of terminology, as it is an index rather than a physical area of usable habitat. Although WUA has units of m^2/m, it represents a physical area only when binary habitat suitability criteria are used (i.e., habitat variables are either suitable (1) or unsuitable (0)). Binary habitat criteria were used for instream habitat analyses prior to 1976 (e.g., McKinley 1957), when Waters (1976) suggested that a sliding scale of suitability from 0 to 1 was more closely related to observed patterns of habitat use (See Section 5). Habitat suitability criteria are or should be developed from relationships between physical habitat and the abundance or presence/absence of aquatic biota, as described in Section 6. When these criteria are applied in hydraulic models, the distribution of habitat suitability through the reach represents the distribution of biota and WUA is an index of abundance or probability of use. Hardy et al. (1983) show that predicted habitat suitability was related to the distribution of fish in a small desert stream and Jowett et al. (1991) demonstrated the relationships between habitat suitability and abundance of benthic invertebrate species in four New Zealand rivers. It is difficult to carry out measurements in the field or laboratory that demonstrate, either negatively or positively, that aquatic populations will respond to flow changes as predicted by WUA and some validation studies are described in Section 10.3. However, it seems logical to assume that aquatic biota will be severely limited by the absence of suitable habitat.

The use of weighted usable area has been criticised because it is possible that that a high WUA could be made up of a large area of sub-optimal habitat. This can be checked easily by examining the way in which the average habitat suitability index (*HSI*) changes with flow. The average habitat suitability index is the habitat suitability score at each point averaged over the reach. Numerically, it is equal to the WUA (m^2/m) divided by the average water surface width.

In the following example, various suitability criteria are applied to a reach to show how habitat/flow relationships should be interpreted. This example assumes that a depth of 0.5 m and velocity of 0.5 m/s are optimum (these are Tennant's trout criteria). The habitat analysis indicates that a flow of 3.5 m^3/s provides maximum WUA and the average habitat suitability index was also highest at 3.5 m^3/s indicating that this flow provides the best habitat quality (Fig. 10.1).

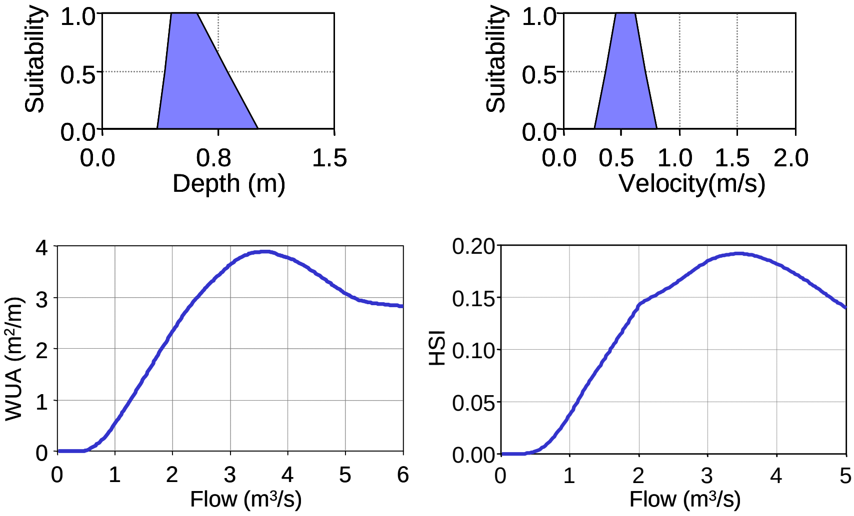


***Figure 10.1*: Variation of weighted usable area (WUA m^2/m) and habitat suitability index (*HSI*) with flow for trout habitat.**

At 5 m^3/s, WUA was still high, but a large area was sub-optimal. When these relationships are displayed as a 2D plot of habitat suitability, it is apparent that the greatest concentration of suitable habitat is at 3.5 m^3/s (Fig. 10.2).

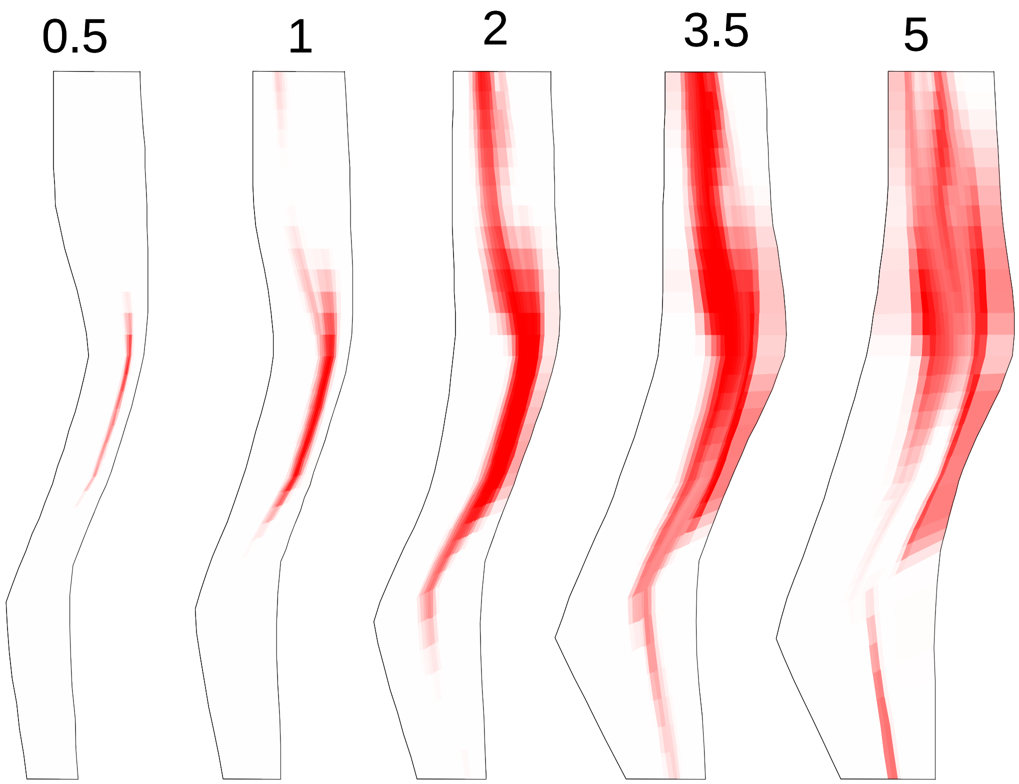


***Figure 10.2*:** **Habitat suitability at flows of 0.5–5 m^3/s, with flow increasing from left to right.**

When a similar analysis was repeated for upland bully habitat, weighted usable area showed two peaks, one at 0.5 m^3/s and a higher peak at 3.5 m^3/s (Fig. 10.3). However, the way in which the average habitat suitability index changes with flow indicates that habitat quality declines with flow. In this case, the area of the stream is increasing as the average habitat quality is decreasing and maximum WUA is created by a large area of sub-optimal habitat. This is easily seen on the 2D plot (Fig. 10.4).

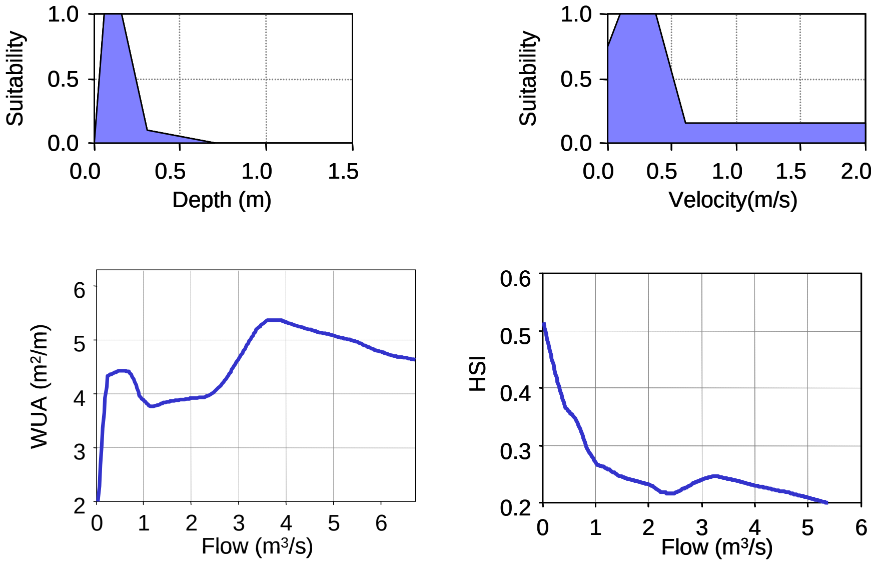


***Figure 10.3*:** **Variation of weighted usable area (WUA) and habitat suitability index (*HSI*) with flow for upland bully habitat.**

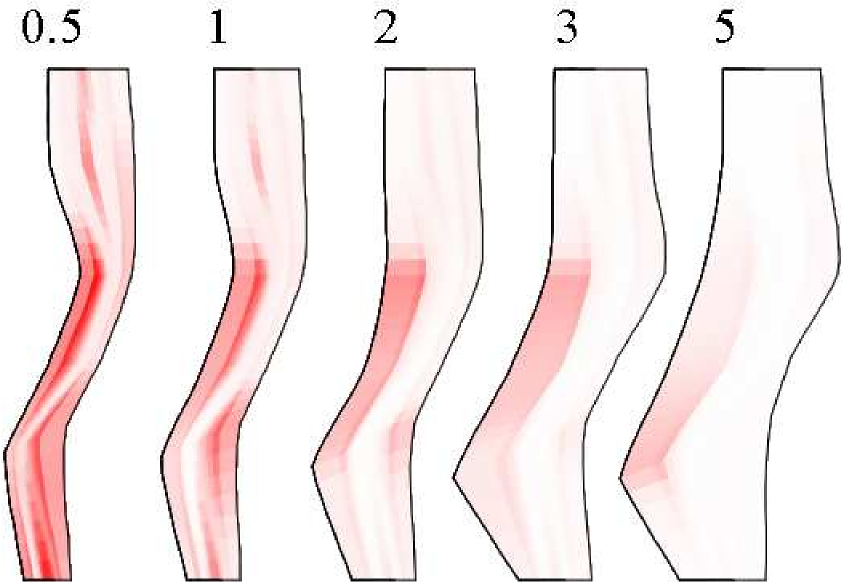


***Figure 10.4*:** **Upland bully habitat suitability at flows of 0.5–5 m^3/s, with flow increasing from left to right.**

10.1.1 Effect of substrate

Substrate size is often taken into consideration when evaluating habitat suitability. The substrate size is controlled by water velocities and depths at channel forming flows and in habitat analyses flows are usually considerably lower than channel forming velocities and the assumption is made that the substrate size will not change. The purpose of defining substrate composition throughout the survey is to calculate depths and velocities over each substrate type and weight habitat suitability accordingly, so that it is possible to calculate the flow that would generate the best possible velocity and depth over the substrate. For example, it is possible to predict the flow that will maximise trout spawning by achieving spawning depths and velocities over gravels.

In practice, substrate composition and its distribution does not influence the shape of habitat/flow relationships, although it is possible to imagine hypothetical cases where it might. This is easily tested by calculating habitat/flow relationships with and without substrate suitability. If substrate is ignored values of WUA will be higher, but the shape of the curve will be the same (Fig. 10.5).

There are some rare circumstances though where substrate, or cover, may influence the shape of a habitat/flow relationship, such as if substrate composition varied systematically across a channel, and, in the case of cover, when the wetted margin recedes from undercut banks. The modeller must be alert to these possibilities when conducting the survey and modelling.

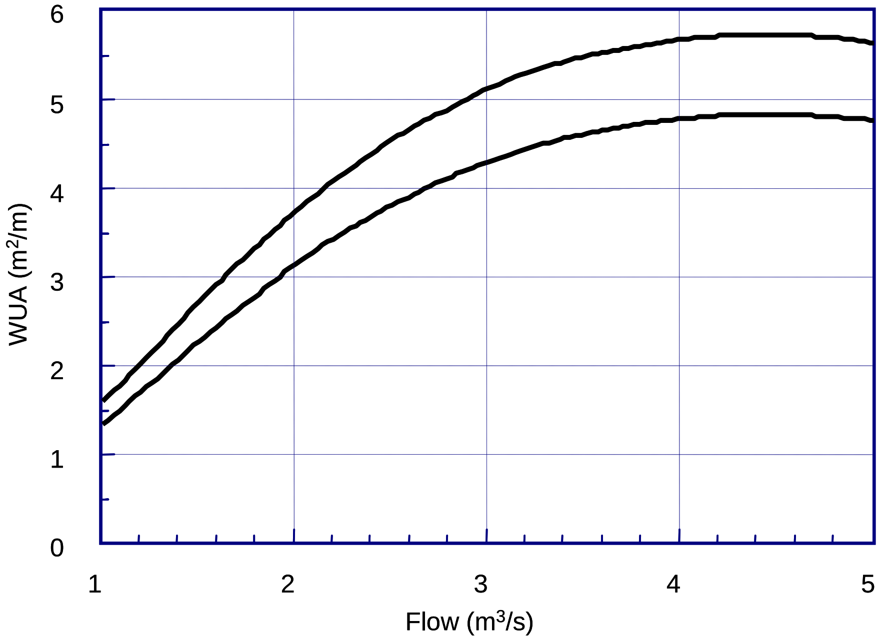


Figure 10.5: **Effect of substrate suitability on brown trout habitat. The upper curve ignores substrate composition the lower curve takes substrate composition into account in habitat suitability.**

10.2 Time series analysis

It is possible to use the relationship between habitat and flow to convert a time series of flows into a time series of habitat, in the same way a stage-discharge rating curve is used to convert a time series of water levels to a flow record. The habitat/flow relationship is not usually monotonic, i.e., there is more than one possible flow for a single value of habitat. However, if low flows are of primary interest, it is possible to use only the low flow side of the habitat/flow relationship for the flow to habitat conversion.

The analyses of a habitat time series are usually similar to hydrological analysis, with habitat duration statistics and seasonal variations. For example, habitat exceedance has been used as a method of setting minimum flows – setting the minimum flow as the flow which maintains habitat at a level that is exceeded for some percentage (e.g., 90%) of the time. However, habitat exceedance and flow exceedance are equivalent if the habitat/flow relationship is monotonic.

Stalnaker et al. (1995) recognise that temporal evaluations are routine in the water management disciplines and argue that a habitat time series presents biological information in a way that is familiar to managers and engineers. This argument does not take the non-monotonic relationship into account, nor does it take into account the fact that managers and engineers should not interpret biological information in the same way as they would interpret hydrological information, as discussed in the following section. While the seasonal variation in habitat and the bottlenecks that may be created are important, this is often more clearly handled by carrying out the appropriate hydrological analyses before converting to habitat. For example, it is much simpler to assess habitat at the mean annual low flow and median flow which are thought to be ecologically relevant flow statistics for trout and benthic invertebrates, respectively (see Section 10.3.1).

10.3 Effect on species abundance

The most difficult and uncertain part of an instream habitat analysis is the biological interpretation of the results. The two key elements are the habitat suitability criteria that are used to calculate habitat and the linkage between available habitat and aquatic populations. These two issues can be discussed and argued without resolution, although the bottom line is that there must always be some suitable habitat if an aquatic species or use is to be maintained.

The often questioned assumption at the heart of habitat based flow assessments is that there is some relationship between the amount of habitat and the abundance of the aquatic species. Species abundance is influenced by factors other than habitat that are not necessarily flow related, so that it is often difficult to demonstrate relationships between species abundance and habitat. However, it is intuitively reasonable to expect that the amount of habitat available would set a limit to population size, in the absence of other limiting factors (i.e., habitat availability would set the outer envelope of abundance for a given population). The factors that influence abundance, but are not related to flow, do not change with flow and need not be considered when assessing the effect of flows changes.

10.3.1 New Zealand studies

Relationships have been established between habitat and brown trout abundance in the form of Jowett's (1992a) trout abundance multiple regression model (see Section 11.1). When data on drift feeding adult brown trout were collected in three New Zealand rivers (Hayes & Jowett 1994) and used to construct habitat suitability curves (Fig. 6.4), there was a significant relationship ($r = 0.395$, $n = 59$, $P < 0.001$) between adult brown trout abundance and percentage suitable habitat in 59 rivers (Fig. 10.6).

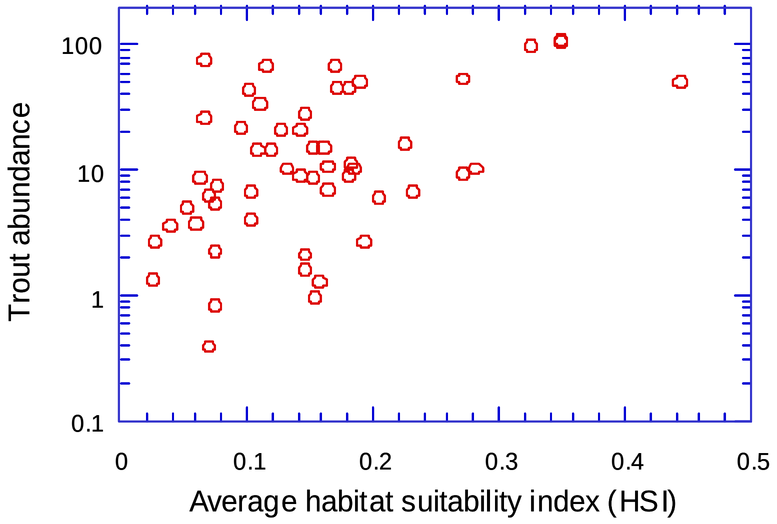


***Figure 10.6*: Relationship between brown trout abundance (number per hectare) and average habitat suitability index for adult brown trout (from data in Jowett 1992a).**

Jowett's trout abundance model is sufficiently well developed that it is now possible to estimate adult brown trout densities that result from alternative water management strategies, but not necessarily year to year variations. A study in the Kakanui River (Jowett 1992a) showed that trout densities could be highly variable from year to year (Fig. 10.7), as a result of intermittent recruitment. However, the distribution of adult trout along the river was related to the distribution of food and habitat. The model C includes the average habitat suitability index (HSI) at mean annual minimum flow and the average habitat suitability index for food producing (benthic invertebrate) habitat at median flow. The other factors in the model usually alter little with flow.

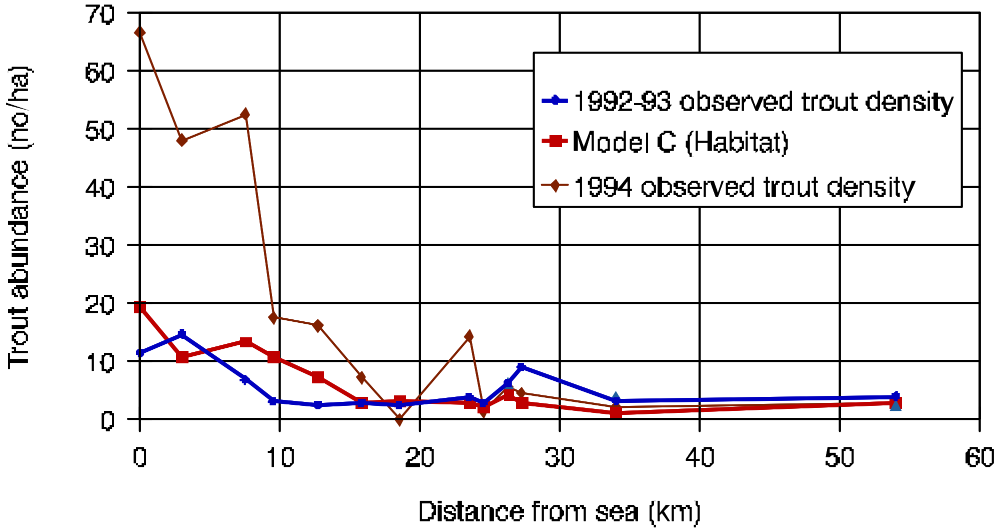


Figure 10.7: **Comparison of predicted trout distribution using Model C (Jowett 1992a) and measured trout distributions in the Kakanui River in 1992–93 and 1994 (from Jowett 1995).**

Relationships have also been established between habitat and; 1) native fish abundance (Jowett et al. 1996a, 2) total invertebrate biomass (unpublished data), and 3) the abundance of some invertebrate species (Jowett et al. 1991; Jowett 1992b). These studies simply show that the various species are most abundant where the habitat is most suitable. This result is to be expected as the habitat suitability curves are derived from the measurements of abundance and hydraulic characteristics. However, they do demonstrate a degree of transferability between rivers. For some species and life stages it may never be possible to establish definitive relationships between the amount of suitable habitat and species abundance. Unless a high proportion of the available habitat is occupied and density dependent mortality or migration occurs, there may not be any relationship between habitat and abundance (i.e., density independent factors may keep populations below levels where habitat becomes limiting). The aim of an assessment of flow requirements is to provide sufficient habitat for the maintenance of all life stages of target species and for the other flora and fauna that make up the food chain in the stream ecosystem, and in this way to maintain the life-supporting capacity of the stream.

Application of native fish habitat preference curves to minimum flow assessments, illustrates how multiple species can be considered. There are usually between 3 and 8 native fish species present in any short section of a New Zealand river. Each species has its own habitat preferences. One group of species is usually found along river margins in shallow slow-flowing water (upland bully, Crans bully, Canterbury galaxias) or in smaller streams. Another group is found mid-stream in the swiftest of water (torrentfish and bluegill bully). Habitat-flow assessments for these two groups would suggest that the edge-dwellers would benefit from extremely low flows, whereas the fast-water species would prefer a flow equivalent to a continuous flood. There is another group of fish that occupies habitats intermediate between these two extremes (common bully, redfin bully). Minimum flows determined on the basis of these intermediate habitat preferences are a compromise that provides some low velocity habitat for the edge-dwelling fish and some high velocity habitat for the fast-water dwellers.

10.3.2 Overseas validation studies

Some comparisons between WUA and fish biomass show poor or negative correlations (e.g., Orth & Maughan 1982; Condor & Annear 1987; Irvine et al. 1987; Zorn & Seelbach 1995; Bourgeois et al. 1996) for some fish species, whereas others have established significant positive relations (e.g., Stalnaker 1979; Nehring & Miller 1987; Nehring & Anderson 1993).

Orth & Maughan (1982) found no significant correlations between weighted usable area and standing stock for adult and juvenile smallmouth bass in any season. However, for the freckled madtom, the central stoneroller, and the orangebelly darter, they found consistently significant correlations between weighted usable area and standing stock during the summer. Zorn & Seelbach (1995) carried out a series of experiments with smallmouth bass, and like Orth & Maughan (1982), could find no correlation between WUA and short-term carrying capacity. Zorn & Seelbach (1995) concluded that instream habitat methods based on velocity requirements may not be appropriate for pool-dwelling species.

Condor & Annear (1987) compared weighted usable area (WUA) to standing crops of trout (*Salvelinus* and *Salmo* spp.) in Wyoming streams. They found no significant correlation for low flow WUA and the measured standing crop among different streams. In contrast, Stalnaker (1979) found a significant correlation between brown trout standing crop and WUA at low flows, in streams with similar water quality and drainage characteristics. Condor & Annear (1987) concluded that significant WUA/standing crop relationships were difficult to determine between streams when standing crops may be influenced by habitat attributes other than depth, velocity and substrate.

Irvine et al. (1987) applied North American habitat suitability criteria for underyearling rainbow trout (Bovee 1978) to small experimental channels in the Waitaki valley and found that WUA did not explain the distribution of underyearling rainbow trout. They suggested that the distribution of young rainbow trout might be better explained by food availability.

Bourgeois et al. (1996) evaluated the relationship between weighted usable area (WUA), predicted by the physical habitat simulation (PHABSIM) model, and the population density of juvenile Atlantic salmon *Salmo salar* in Catamaran Brook, New Brunswick, Canada. The PHABSIM model was applied to 19 sites, representing four habitat types. Few positive, significant relations were established between Atlantic salmon density and WUA; r^2 values ranged from 0.18 to 0.95, with the best relations occurring at the scale of habitat type (5 of 16 comparisons were significant, $P < 0.05$). The WUA values calculated from the 15-day average flow before fish sampling displayed the best associations with fish density.

Studies in the Black Canyon of the Gunnison River, Colorado over the period 1981–1986 (Nehring & Miller 1987; Nehring & Anderson 1993) showed that recruitment of rainbow and brown trout at age 1+ was almost totally controlled by fry habitat availability (expressed as fry WUA) in the first 30–45 days post emergence. Fry WUA was in turn controlled by spring-early summer discharge patterns (i.e., high discharge created habitat that was unsuitable for fry). Brown trout biomass was positively correlated with average summer adult brown trout WUA (April through October) over the 6 years of study. This indicates that average adult brown trout WUA was a good predictor of brown trout density and biomass, regardless of whether the stream is at carrying capacity, as has been suggested is necessary for the establishment of a relationship between WUA and abundance (Bovee 1982; Orth & Maughan 1982; Mathur et al. 1985). There was no significant correlation between either adult rainbow trout density or biomass and average adult rainbow trout WUA. Brown and rainbow 1+ density and biomass was not correlated with parent spawner density or spawning WUA.

Nehring & Anderson (1993) used PHABSIM to investigate the effect of flow related habitat changes on rainbow trout and brown trout population in Colorado streams over a long term time series (13 years). Critical habitat limiting periods (bottlenecks) were identified for newly emergent fry, egg incubation, and spawning in 10 of 11 streams studied. They concluded that instream habitat analysis had proven valuable in interpreting the relationship between flow and recruitment, and that the results of the study largely verify the basic assumption of IFIM, i.e., there is a positive relationship between WUA and fish standing stock. Their experience with the Gunnison, and seven other streams in Colorado, is that it is the 2–4 week old fry stage that is most often the bottleneck in Colorado trout streams. High gradient streams in Colorado have a limited amount of shallow, low velocity habitat which trout fry require. This habitat is further limited by the magnitude and duration of the spring run-off, which coincides with the period of rainbow and brown trout fry emergence.

11. Effectiveness of habitat-based flow assessments

New Zealand experience with habitat and flow assessments

- Adult brown trout density is related to average habitat suitability indices for adult brown trout and food producing habitat.
- Flow recommendations based on instream habitat assessments have been successful and biological response was as expected in six out of seven cases described here.
- The flow regime in some of these rivers was far from “natural”, particularly in the Waiau and Monowai rivers, yet these rivers contain excellent trout and invertebrate populations.
- These case studies do not support commonly held views that more flow is better and that all aspects of a natural flow regime are important – the ‘natural flow paradigm’.
- However, floods are important for flushing out accumulations of periphyton and fine sediment in some cases allowing fish migration.

11.1 Brown trout model

Using data collected for the ‘100 rivers survey’, Jowett (1992a) developed a model of the abundance of large brown trout in New Zealand rivers (Section 10.3.1). Weighted usable area for trout habitat, space, and WUA for food production, food, plus seven other variables explained 87.7% of the variation in numbers of large brown trout in 59 New Zealand rivers. The most important variables were WUA% (equivalent to HSI) for trout habitat, WUA% for food production, instream cover, and water temperature as an overriding factor. Other significant variables included percent sand substrate, % area of lakes in catchment, elevation, gradient, and percentage of the catchment developed for agriculture. Sand substrate is very poor food producing habitat and it is rare to observe brown trout in areas where the predominant substrate is sand; lake outlets are well known for their high trout stocks; headwaters usually contain lower trout densities than the lower reaches of a river; trout populations in high gradient rivers are severely depleted by floods (Jowett & Richardson 1989); and pastoral development appears to have an adverse impact on trout.

Perhaps the most interesting concept in the brown trout model is the flow at which the instream habitat variables (HSI) are calculated. In a natural river, flow and habitat vary with time. The quality of habitat was calculated at three flows; mean annual low flow, median flow, and mean flow. The quality of adult trout habitat at mean annual low flow was more closely related to trout numbers than the habitat available at the higher flows. This suggests that the quality of trout habitat at low flow is one of the limiting factors in the system – a kind of bottleneck. The quality of habitat for food production (benthic invertebrate habitat) at median flow was more closely related to trout numbers than the amount at either low or mean flow. Thus, it appears that even if there is adequate habitat at low flows, a trout population is likely to be controlled by the food producing capacity of the river at median flow rather than the capacity during more extreme events. Ideally, the food producing capacity should be derived by integrating the amount of habitat over the full flow regime of the river; however, habitat at median flow appeared to be a reasonable estimate.

Although methods of assessing flow requirements continue to be developed and debated, very few studies examine how well modified flow regimes have achieved their desired outcomes. Armour & Taylor (1991) surveyed 35 U.S. Fish and Wildlife field offices that had been involved in 616 IFIM applications of which 6 had follow up monitoring, the results of which were not reported. The survey found that opinions on IFIM were divided, with 40% considering the method technically too simplistic, 41% considering it too complex to apply, and 9% considering it not acceptable or biased. However with any flow assessment method, the critical test is whether it is successful in achieving the desired outcome. In the U.S. survey, half of the respondents rated success as higher than neutral, while one-third rated it lower.

11.2 Case studies of biological response to flow change

We review six New Zealand case studies involving trout, benthic invertebrate and indigenous fish communities where minimum flow and flow regime recommendations have been made and implemented, and examine the available biological data to determine whether these recommendations have been successful in achieving their desired goals.

In each case, instream habitat surveys were carried out in study reaches that were most affected by flow change. The instream habitat surveys were either closely spaced cross-sections in representative reaches or cross-sections selected by a stratified-random sampling method (habitat mapping). Instream habitat modelling was carried out using RHYHABSIM (Jowett 1989; Clausen et al. 2004). Minimum flow recommendations were made after examination of the habitat (WUA)/flow curves. Usually, flow requirements were determined from a breakpoint determined by drawing a horizontal line through the maximum and extending a line through the low flow section of the curve (Fig. 5.2).

Recommendations for extra flow releases as a means of flushing deposits of fine sediments or accumulations of filamentous algae were made where we considered it necessary and beneficial. The magnitude of these releases was calculated by the method of Milhous (1998), as implemented in RHYHABSIM (Section 7.8).

The biological response to flow changes were collected from a variety of sources, such as drift-diving counts of trout, benthic invertebrate sampling, angler surveys, and electric fishing surveys. In most cases, biological data were not collected specifically for the evaluation of flow changes. Where appropriate, trout densities in rivers with modified flow regimes were compared to national drift diving survey data (Teirney & Jowett 1990) to show the relative magnitude of the response and how closely the trout density approached national maxima in unmodified rivers.

11.2.1 Tekapo River

The first study of instream habitat carried out in New Zealand was in the Tekapo River (Jowett 1982), where diversion of flow for the Waitaki Power Development in 1978 had reduced the flow from a mean of 80 m^3/s to zero. Although no minimum flow is provided below the diversion structure, tributary flows increase the mean flow to about 12 m^3/s in the lower section of river about 45 km downstream of the diversion. The habitat analysis of the lower section of river (Jowett 1982) showed that a flow of about 10–13 m^3/s provided maximum trout spawning and food producing habitat for that river.

Spawning surveys were carried out by MAF Fisheries before and after diversion reduced the flow. In July 1974, prior to diversion 54 fish and 10 redds were counted. In September 1974, 35 fish and 24 redds were counted. After diversion, a survey in June 1978 counted 250+ fish and 100+ redds (Jowett 1978).

There were no records of trout density prior to diversion, when the Tekapo River was hardly recognised as an angling river and it is not mentioned in angling surveys of that time (Allan & Cunningham 1957; Graynoth & Skrzymski 1973a). Trout populations were surveyed by drift-diving in 1986 & 1989 (Teirney & Jowett 1990) and densities of 94–240 brown and rainbow trout (> 20 cm) per km were recorded (Fig. 11.1). It is now one of the most popular rivers in the region, with angler use of 2400 days in 1994/95 and 4900 days in 2002/02 (Deans et al. 2004).

The diversion of turbid Lake Tekapo water increased water clarity in the river, and increased the variability of flow relative to mean flow because the source of water is unregulated tributary streams. Naturally occurring flows in the Tekapo River were probably too swift to support a good trout fishery, but the diverted turbid water now flows at a lower gradient through the Tekapo Canal, which has become the sixteenth most popular angling ‘river’ in the country (Deans et al. 2004).

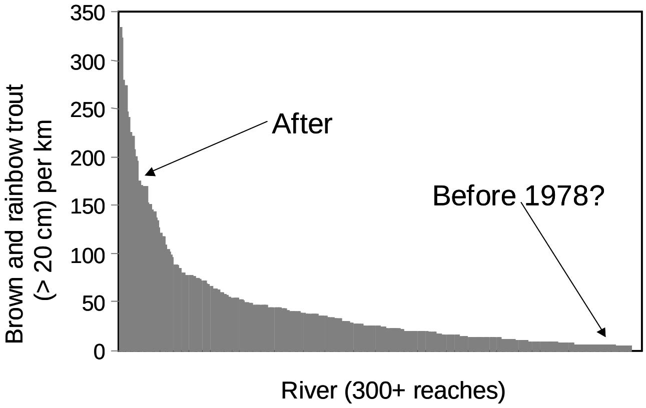


***Figure 11.1*: Total numbers of brown and rainbow trout (> 20 cm) per km in the Tekapo River before and after a reduction in flow from about 92 m^3/s to about 12 m^3/s compared to ranked national data from 300+ river reaches.**

11.2.2 Waiau River

Practically all of the natural mean flow of about 450 m^3/s of the Waiau River in Southland was diverted through the Manapouri Power Station between 1977 and 1995. Tributary flows increased minimum flows in a 20 km section of river immediately below the diversion structure, from 0.3 m^3/s immediately below the structure to about 3 m^3/s about 20 km down stream.

Instream habitat surveys were carried out in a 20 km section of river immediately below the diversion structure. The instream habitat analysis (Jowett 1993c) indicated that a flow of 12 m^3/s or greater would provide excellent brown trout habitat and a minimum flow regime of 12 m^3/s in winter and 16 m^3/s in summer was consequently implemented.

Drift-diving trout surveys carried out over the 20 km section of river before and after the minimum flow was increased show that numbers of brown and rainbow trout (> 20 cm) increased (Fig. 11.2) about four-fold (Moss 2001). The trout fishery in this river is now regarded as excellent, with good numbers of trout and anglers, and high catch rates. In the 1994/95 fishing season just before the implementation of the minimum flow, angler usage in the whole river (c. 90 km) was 7700 days and this increased to 14600 angler-days in 2001/02 (Deans et al. 2004) against the national trend of an overall decline in river fishing. By 2004 the Waiau had become the eighth most popular trout fishing river in New Zealand, as it was before diversion (Allan & Cunningham 1957), and the 2004 trout densities in the Waiau River ranked it as one of the top rivers in New Zealand (Fig. 11.3). In 2004 the invasive alga, didymo (*Didymosphenia geminata*), became established in the Waiau River and there is concern that the fishery may deteriorate.

Although river flows have reduced from 450 m^3/s to 12–16 m^3/s, there is no evidence of any detrimental effects on native fish probably because there are no indigenous species that are found solely or predominantly in large rivers (McDowall 1994). However, the visual appearance of the river has changed with a loss of the 'large river' character.

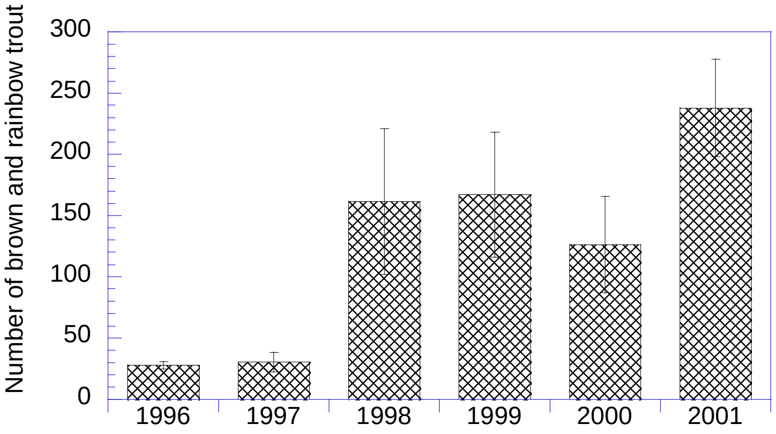


***Figure 11.2*: Numbers of large brown and rainbow trout (> 20 cm) in the Waiau River per km before and after the (1997) implementation of a minimum flow regime of 12 m^3/s in winter and 16 m^3/s in summer.**

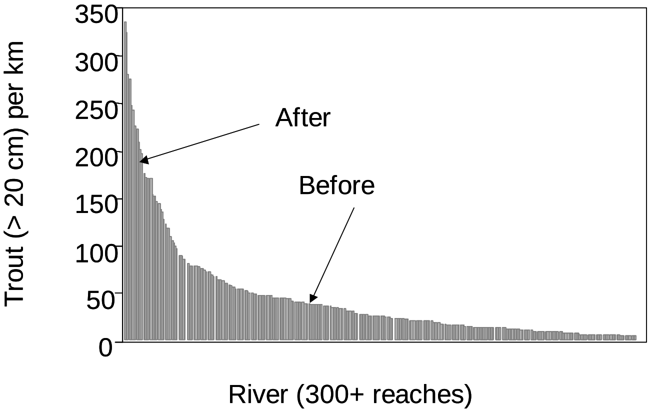


***Figure 11.3*: Total numbers of brown and rainbow trout (> 20 cm) per km in the Waiau River before and after an increase in flow from about 0.3 m^3/s to about 12–16 m^3/s compared to ranked national data from 300+ river reaches.**

The operation of the hydro-electric power development results in regular flood releases of high flows (100–200 m^3/s) and these are supplemented by releases of 45 m^3/s four or five times each summer for recreational purposes. However, additional releases to flush filamentous algae were advised on a 'when needed basis'. Subsequent monitoring, pre didymo, showed that additional flushing (of native algae) was not necessary unless the spring/early summer spill flows do not occur.

11.2.3 Monowai River

In some cases, the primary objective of flow recommendations has been the maintenance of healthy benthic invertebrate communities. Lake Monowai is regulated for hydropower and flows from this lake varied frequently from near zero to full generation (20 m^3/s), depending on electrical demand. In 1995, the minimum flow in the Monowai River was increased from about 0.2 m^3/s to 6 m^3/s to provide habitat for benthic invertebrates (Jowett & Biggs 2006), but flows still frequently vary from 6 m^3/s to 20 m^3/s.

As part of a national annual benthic invertebrate survey (Scarsbrook et al. 2000), seven pooled 0.1 m^2 Surber samples were collected in run habitats (water depth 02–0.4 m, velocity 0.6–1.0 m/s) before and after the increase in minimum flow. This showed that the increase in minimum flow doubled benthic invertebrate densities (from 310 per m^2 to approximately 650 per m^2) and taxon richness (from 8 to an average of 17) (Fig. 11.4).

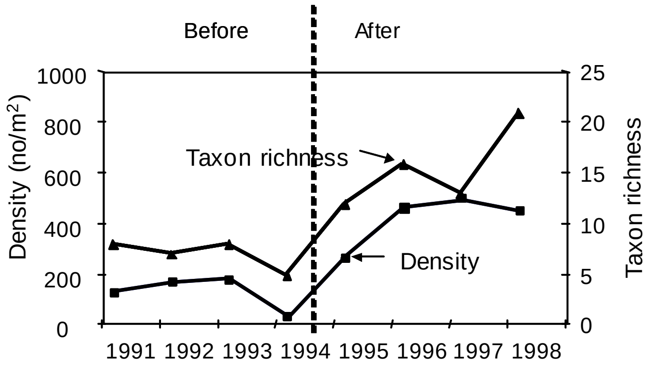


***Figure 11.4*: Density and taxon richness of benthic invertebrates in the Monowai River before and after implementation of a minimum flow control (Jowett 2000).**

11.2.4 Moawhango River

The Tongariro Power Development diverted the natural flow of the Moawhango River, a mean flow of about 9.6 m^3/s, to the Tongariro River, leaving practically no flow in the Moawhango River below the dam. As part of the re-licensing process, a minimum residual flow of 0.6 m^3/s was suggested and implemented to re-establish benthic invertebrate communities below the dam (Jowett & Biggs 2006).

Benthic invertebrates were sampled before and after the minimum flow was implemented. In 2002, after about a year with a minimum residual flow in the Moawhango River, the composition of the invertebrate community had changed considerably. The proportion of the invertebrate community composed of mayflies + stoneflies + caddisflies (%EPT: a measure of the relative abundance of 'healthy' invertebrates) increased from 37% to 57% with the increase in flow, to the extent that it is now similar to the 60% EPT composition in the river upstream of the dam (Fig. 11.5). For all taxa, the relative abundance of 'target' taxa increased with the increase in flow, whereas the relative abundance of 'non-target' taxa decreased (Fig. 11.6).

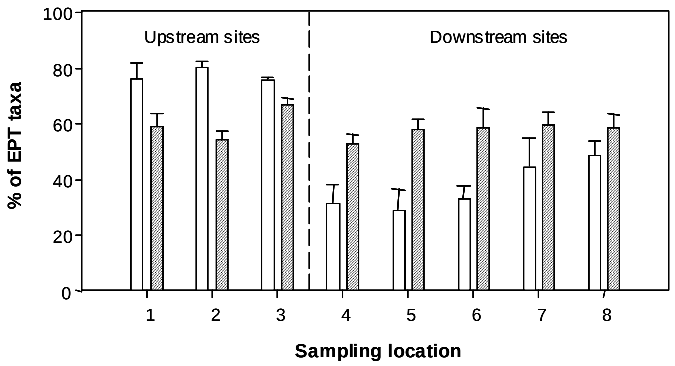


***Figure 11.5*: Total percentage of EPT taxa (mayflies + stoneflies + caddisflies) (x + 1 se, n = 5) at the sampling sites at locations upstream and downstream of the Moawhango Dam, collected in 1997 (open bars) and in 2002 (hatched bars) (Jowett & Biggs 2006).**

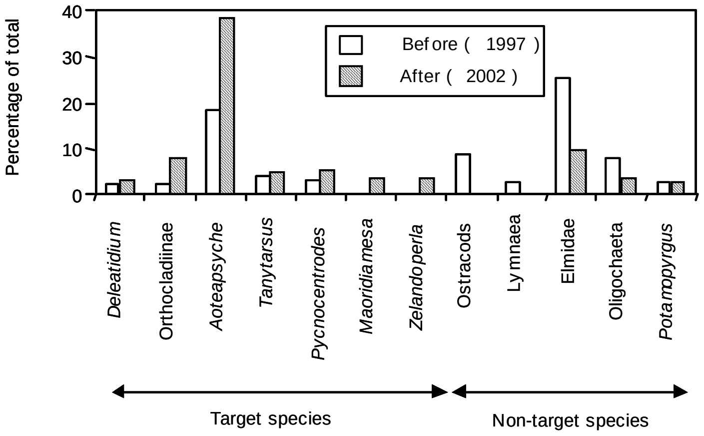


***Figure 11.6*: Benthic invertebrate species composition in Moawhango River before the minimum flow was implemented (open) and after (hatched), showing how dominance of 'target' invertebrate species (left) increased and 'non-target' invertebrate species (right) decreased as a result of a flow increase.**

Subsequent to the invertebrate monitoring described above, flushing flows were recommended for the Moawhango River, because spill flows from the dam were relatively infrequent and sediment and periphyton was accumulating in the lower reaches. The flushing flow analysis (Fig. 11.7) showed that a flow of 20 m^3/s would cleanse more than 80% of the base flow streambed and disturb less than 20% of the armour layer. Tests showed that flushing flows of 20 m^3/s were effective (Fig. 11.8) and these have been implemented and are expected to result in further improvement to the invertebrate community.

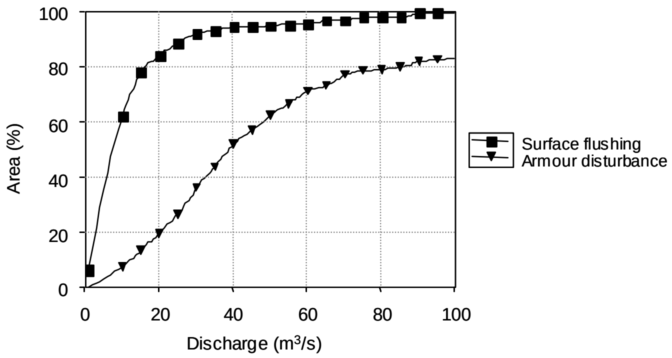


***Figure 11.7*:** **Percentage of the Moawhango River bed area flushed by flows of 0 to 100 m^3/s calculated using the Milhous (1998) method.**

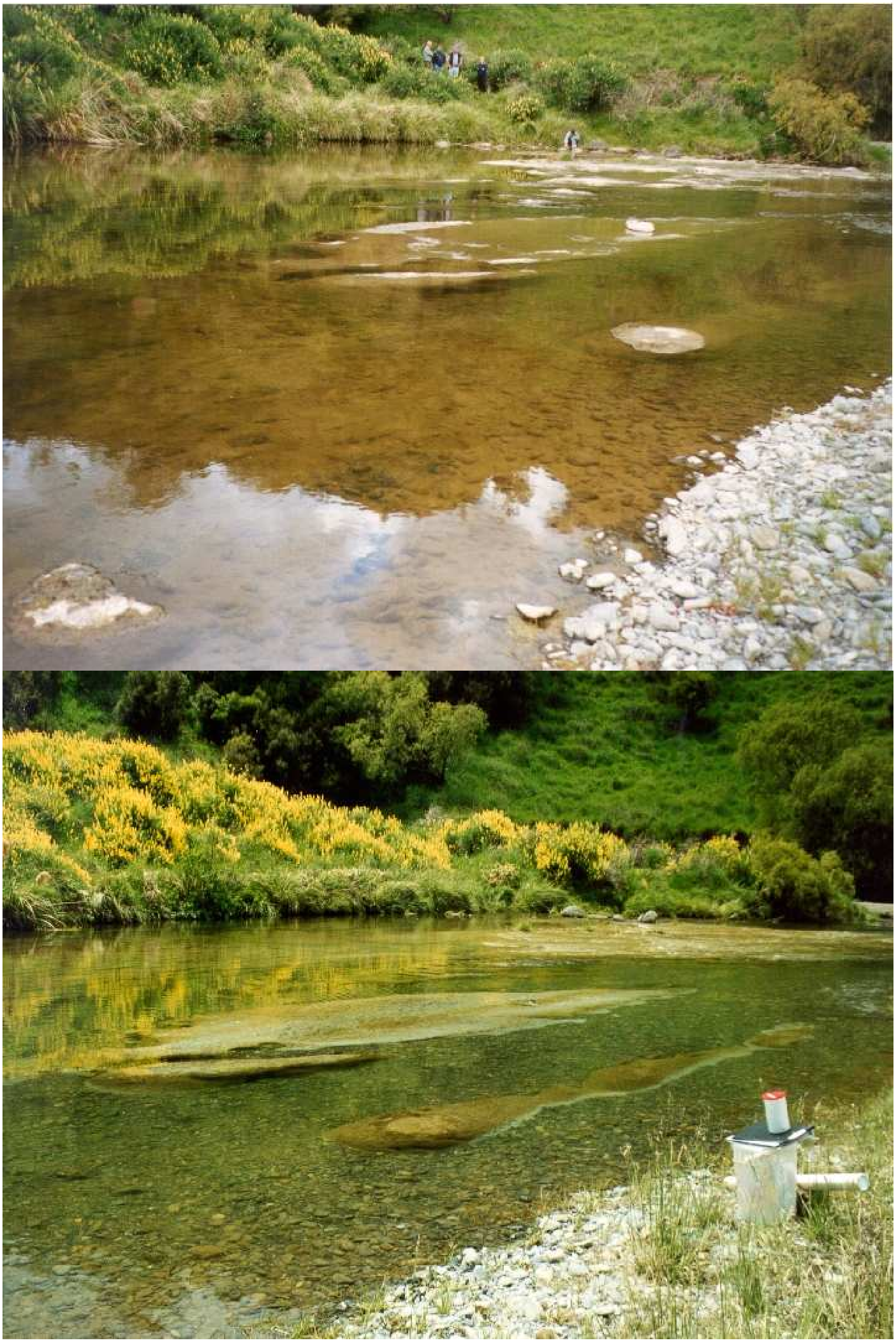


Figure 11.8: **Moawhango River after 8 months of nearly constant flow (above) and 7 days after a flushing flow of 20 m^3/s (below).**

11.2.5 Ohau River

The only case where a prediction of the likely response of trout abundance to flow management based on the quality of habitat was not successful is in the Ohau River in the South Island. The mean flow in the Ohau River was 80 m^3/s prior to diversion for hydropower development in 1979, after which less than 1 m^3/s was left in the river. The Ohau River flows from Lake Ohau, which has a low dam and structure to release water. It then flows for a short distance before it enters the artificially created Lake Ruataniwha. There is no flow in the river below Lake Ruataniwha. An instream habitat survey (James et al. 1992) showed that a flow of 10 m^3/s would provide excellent trout habitat and this flow has been released at the lake outlet since 1994. Although the river now provides what is regarded as excellent angling water and trout habitat, trout numbers and angler usage have remained low, with 636 anglers-days in 1994–95 (Unwin & Brown 1998) and 500 angler-days in 2001–02 (Unwin & Image 2003). The present low numbers of trout in this section of river may be related to problems with recruitment and fish passage between Lake Ohau and the river, or simply a preference for the environment in Lake Ruataniwha. No flushing flows were recommended because there was no facility for controlled release of high flows.

Reports on the state of the trout fishery do not distinguish between the upper section of the river and the lower section above Lake Benmore. Prior to the formation of Lake Benmore in 1965, there is little mention of the Ohau River in angling literature. An early angler survey by Allan & Cunningham (1957) does not mention the river. Brian Turner (2002) describes fishing in the Ohau River before the creation of Lake Benmore, from just below the State Highway bridge to the lake, and gives graphic descriptions of fishing for large fish in fast water, catching an average of two per day. The Ohau river appears to have become more popular with the creation of Lake Benmore. In terms of crop, Graynoth & Skrzymski (1973b) ranked it fifth of seven angling rivers in the Waitaki Valley district. The 1976 Annual Report of the Waitaki Acclimatisation Society described that Ohau River as one of the finest angling streams in the district. Jowett (1978) described the Ohau River as having fast turbulent flow with relatively low numbers of large fish attracting experienced enthusiastic anglers. He also notes that the river was being heavily fished near Lake Benmore, with high numbers of rainbow trout and moderate numbers of brown trout.

11.2.6 Onekaka River

The Onekaka River is a relatively small river (16 km^2 catchment area) that enters Golden Bay midway between Takaka and Collingwood. A 10.7-metre dam was built in the headwaters in 1928 to supply power to an ironworks blast furnace that operated until 1952. In 2003, a small hydroelectric scheme was built using the head created by the old ironworks dam. The powerhouse discharges water back into the Onekaka River about 800 m below the dam. A minimum flow of 20 L/s was required below the dam, with a minimum of 50 L/s in the river below powerhouse (Fig. 11.9). The hydroelectric scheme operates with twice daily peaking. One major tributary enters the Onekaka between the dam and power station and contributes about 15% of the mainstem flow.

Habitat suitability curves were developed for the three most common species (koaro, redfin bully and longfin eel) by measuring the physical habitat at a wide variety of 3–4 m^2 quadrates within the Onekaka and other rivers. An instream habitat survey was carried out to determine minimum flow requirements (Richardson & Jowett 1995). The fish population in the Onekaka was assessed by electric fishing 3 x 30 m reaches over a 650 m section of the river. The same three reaches were fished in March–April 2003 (before the power scheme began operating), and in 2004 to 2007. The lowermost reach was located 390 m below the power station discharge at the tributary confluence, the middle reach about 60 m below the power station discharge, and the top reach 260 m above the power station discharge. The abundance of native fish was measured by multiple pass electric fished in permanently marked reaches below the dam and power house for one year before the commissioning of the power scheme and for four years after.

Before commissioning, the median flow in the Onekaka River was 218 L/s and the average annual low flow was about 66 L/s. After commissioning, the median flow below the dam was 30 L/s and the average annual seven-day minimum flow was 20 L/s. However, the minimum flow in the river below the power station was probably little different to what it was before the power station began operating, although the water level typically fluctuated by about 100 mm per day (Fig. 11.9).

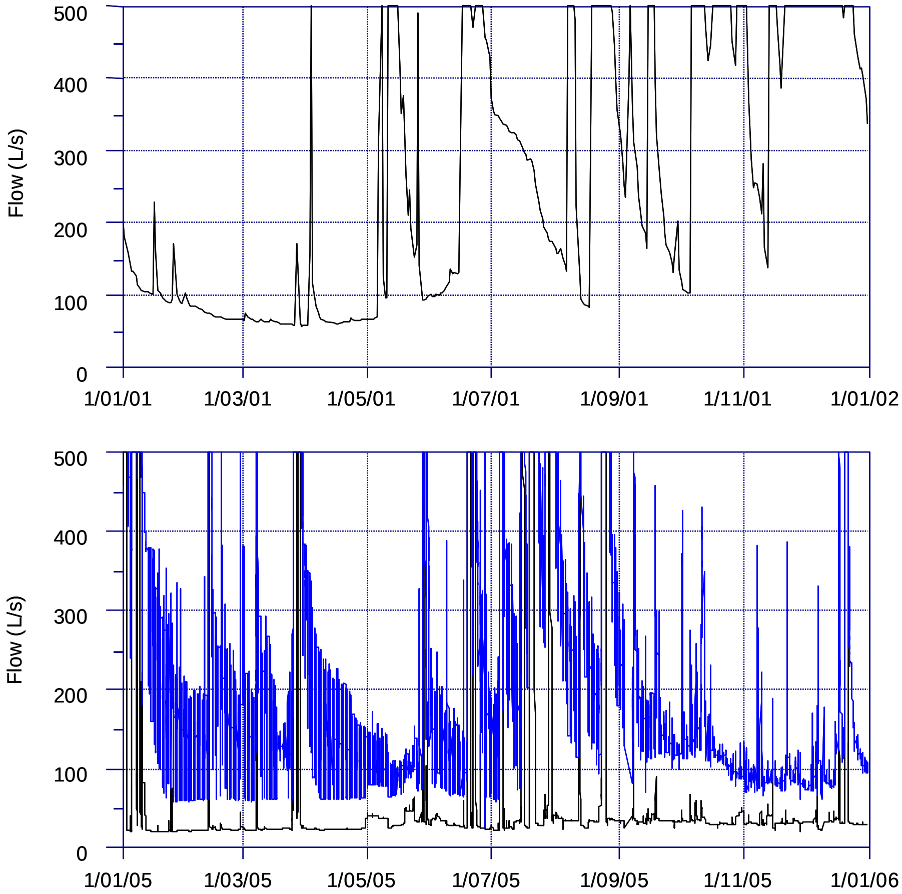


***Figure 11.9*:** **Onekaka River flow in 2001 and 2005. In the lower graph, the upper (blue) line shows flow in river below powerhouse and the lower (black) line shows flows below dam.**

In the survey reach of river below the dam, the number of koaro reduced by 80% from 38 to an average of 7.5 (Table 11.1), There was a 61% reduction in koaro low flow habitat when the seven-day annual minimum flow reduced from 66 to 20 L/s (Fig. 11.10) and a 76% reduction in habitat at median flow (218 L/s reduced to 30 L/s). Koaro are found in fast flowing water (Fig. 11.11) and the response of this species to the flow change was similar to that observed for fast water species in the Waipara River (Section 11.8; Jowett et al. 2005).

Longfin eels are usually considered to be generalists with fairly broad preferences. Nevertheless, longfin eel numbers reduced by 52% compared to a 33% reduction in low flow habitat and 44% reduction in habitat at median flow (Fig. 11.10).

Redfin bully numbers were low and variable over the sampling period, with no obvious reduction in numbers despite a 40% reduction in habitat (Fig. 11.10).

Table 11.1: **Estimated number of koaro, longfin eel and redfin bully in the survey reach of the Onekaka River below the dam. The power station was operating 2004–2007.**

Year	Koaro	Longfin eel	Redfin bully
2003	38	16	1
2004	12	5	4
2005	8	9	5
2006	3	8	2
2007	7	9	1

Below the power station, the magnitude of median and low flows did not change substantially, although there were daily flow fluctuations of 200 L/s decreasing to 100 L/s in mid 2005. Koaro and longfin eel numbers initially decreased slightly, but recovered to initial levels by 2006-2007 (Table 11.2). Redfin bully numbers declined and did not recover. There may have been some movement of redfin bullies into the upstream reach where flows were more stable.

Habitat rather than floods probably caused changes in fish numbers. Before the station was commissioned (1999–2003 data), there were 14 large floods per year on average in the Onekaka, with an average annual maximum daily mean flow of 3754 L/s. In 2004–2006, there were 12 large floods per year on average, with a maximum daily mean flow of 4373 L/s.

Table 11.2: **Estimated number of koaro, longfin eel and redfin bully in the survey reaches of the Onekaka River below the powerhouse. The power station was operating 2004–2007.**

Year	Koaro	Longfin eel	Redfin bully
2003	12	32	25
2004	8	19	13
2005	10	21	6
2006	11	38	1
2007	13	32	5

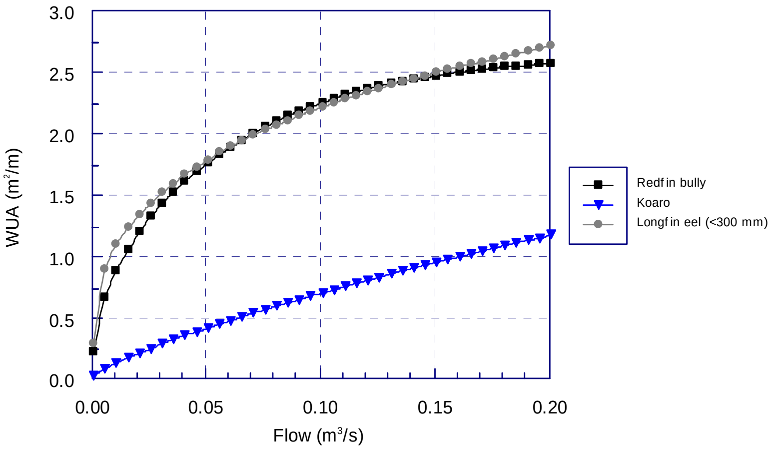


Figure 11.10: **Predicted change in weighted usable area (WUA) for three common species in the Onekaka River.**

***Figure 11.11*: Koaro habitat in the survey reach below the dam.**

11.2.7 Waipara River

The Waipara River is a small gravel bed river and is under considerable pressure from irrigation abstraction. An instream habitat analysis was carried out in 1994 and a minimum flow of 120 L/s was recommended for the maintenance of indigenous fish biodiversity values (Jowett 1994). This flow was based on consideration of habitat for common bullies (*Gobiomorphus cotidianus*), a species with habitat preferences that were intermediate between the fast-water species, torrentfish (*Cheimarrichthys fosteri*) and bluegill bullies (*G. hubbsi*), and the edge-dwelling species, upland bullies (*G. breviceps*) and Canterbury galaxias (*Galaxias vulgaris*), as suggested by Jowett & Richardson (1995).

Following the flow assessment, there was a 3 year study of fish in the Waipara River to determine the effects of flow and flow regime on fish populations (Jowett et al. 2005). Fish populations were surveyed seasonally by electro-fishing eight reaches along 20 km of river for three years. In the first December to May (inclusive) summer (1998–99), flows were extremely low, but were relatively normal in the following year (Fig. 11.12).

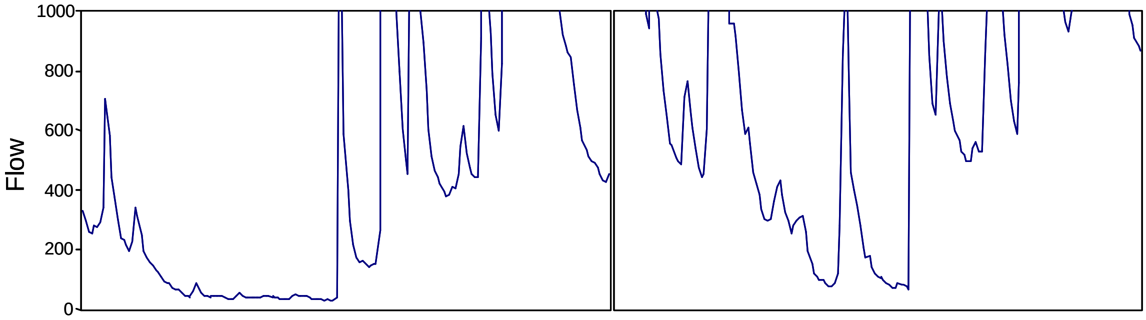


Figure 11.12: **Waipara River flows (L/s) at White Gorge over the dry Dec 1998 to May 1999 summer (left) and the normal Dec 1999 to May 2000 summer (right).**

The fish surveys showed that the effect of low flows on fish populations increased with the magnitude and duration of low flow. In the first summer (1998/99) when the mean flow at White Gorge was 647 L/s, flows were less than 110 L/s for 34% of the time, and fell to a minimum of 31 L/s. These low flows led to a substantial decline in the abundance of 3 of the 4 common indigenous fish species in the river (Fig. 11.13 left). The following summer (1999/00) when the mean flow was 1069 L/s, flows were less than 110 L/s for only 10% of the time, with a minimum of 62 L/s. These conditions resulted in little change in indigenous fish abundance (Fig. 11.13 right).

These results support the recommended minimum flow, and even suggest that the minimum flow recommendations for these indigenous fish species may have been unnecessarily high. However, recent concurrent gaugings carried out by Environment Canterbury and NIWA show that there are flow losses and gains along the river between the White Gorge recorder and the lagoon. In particular, there appears to be a sharp increase in flow below the Omihi confluence as the rivers becomes more confined and a section of severe water loss just above the lagoon, with no flow when the flow at the Teviotdale recorder is about 350 L/s. Concurrent flow measurements carried out during NIWA's study of the Waipara indicate that, on average, the flow at Teviotdale is about 60% higher than the flow at the White Gorge recorder site. At low flows, this relationship indicates that the flow at Teviotdale would be unlikely to fall below 100 L/s, and that in the first summer (1998/99) when flows at White Gorge fell to 31 L/s, the flow at Teviotdale was probably about 154 L/s. These low flows (summer 1998/99) led to a substantial decline in the abundance of 3 of the 4 common indigenous fish species in the river. The following summer (1999/00), flows at White Gorge were less than 110 L/s for only 10% of the time and the flow at Teviotdale probably fell to about 193 L/s and there was little change in indigenous fish abundance. In the final year of the study (2000/01), the flow at White Gorge fell to about 47 L/s, an estimated 174 L/s at Teviotdale, and torrentfish and bluegill bully numbers were affected, although not as much as in 1998/99.

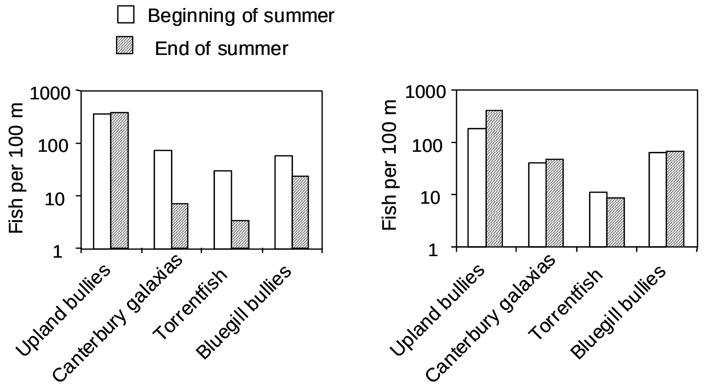


***Figure 11.13*: Indigenous fish abundance in the Waipara River (upper and lower reaches) at the beginning and end of a dry 1998/99 summer (left) and wet 1999/2000 summer (right). Upland bullies and Canterbury galaxias are mainly in the upper reaches and torrentfish and bluegill bullies are mainly in the lower reaches.**

This study demonstrated the resilience of the indigenous fish community, as it redeveloped strongly after the first year of the study even though it had been severely affected by low flows. Some large floods during the study that caused extensive disturbance of bed materials had little effect on fish abundance, and diadromous species (torrentfish and bluegill bullies) were dependent on spring floods opening the mouth for recruitment. Low flows were more detrimental to the fish community than floods, with prolonged low flows reducing the abundance of fish species that prefer high water velocities, and favouring those that prefer low velocities. During periods of low flow, proportionally more fish were found in riffles than runs. This implies that riffle habitat is important in the maintenance of fish stocks and biodiversity during periods of low flow. The key elements of the flow regime were the magnitude and duration of low flows, as well as the occurrence of spring floods that allowed recruitment of diadromous species.

12. Conclusion

Although there have been criticisms of habitat methods, and of IFIM in particular, most of these criticisms have been made by people with little practical experience in the application and interpretation of results. The examples presented here show that the processes that control river morphology result in strong longitudinal hydraulic similarity, i.e., over sections of river the depth and velocity distributions and shape of habitat/flow relationships are remarkably similar. Moreover, when flow is standardised by channel width these characteristics are similar between rivers of broadly similar type, as shown by generalised habitat studies. The selection of reaches is neither a complicated nor sensitive task and the variation of flow with habitat can be determined from relatively few cross-sections. In fact, habitat analyses based on simple hydraulic geometry, 1D surveys or 2D surveys will produce useful and similar results. However, the tasks of survey, calibration, habitat suitability and analysis, and finally the interpretation of results require a good knowledge of river mechanics, hydraulics, and ecology. Survey (habitat mapping) and hydraulic calibration used in RHYHABSIM are relatively robust, but more complex modelling techniques, such as water surface profile modelling and 2D modelling, are more difficult and can be done poorly, both in the execution of the survey and hydraulic analysis.

While many fish and stream insects undoubtedly make use of habitat on a micro scale, many of the features that create microhabitat, such as substrate, bed, and bank forms, vary little with flow and a flow requirement that provides suitable macro-habitat should also provide suitable microhabitat. Instream habitat methods, although often described as microhabitat, are in fact evaluating mesohabitat. The survey techniques described here are capable of predicting depths and velocities to the scale of the survey, which is usually measurements spaced at 0.1–3 m. They do not predict micro-scale hydraulics. Similarly, many habitat suitability observations describe mesohabitats – the characteristics of the area in which the organism lives, rather than the micro-hydraulics of its precise location. In assessing suitability for one target species, we are often assessing conditions for a number of species that live in that area. Riffle-dwelling fish and invertebrates are an example, where the habitat suitability curves describe riffle conditions, rather than microhabitat of the location of an individual organism. The selection of critical values and target species (fish or invertebrate) as an indicator of stream health is a concept that can be applied to flow assessment.

The derivation and use of habitat suitability models (habitat suitability curves) are the most important aspects of flow evaluation. Habitat suitability curves can be derived and used inappropriately. Although habitat suitability criteria are available for many New Zealand aquatic organisms, they can be improved by collecting more data and recalculating habitat suitability models. The question of hydraulic scaling, or transferability between rivers of different size, for benthic invertebrate and rainbow trout habitat is a problem that has yet to be solved.

Although the functional role of flow regime components, such as low flow, flushing flows, and channel maintenance flows, is known, we do not know the degree to which the frequency and duration of these events affect biota, and do not have any method of assigning acceptable frequencies and durations, other than mimicking nature.

Finally, hydraulic habitat modelling is a tool to assist in the decision making process. No flow will maintain maximum habitat for all aquatic organisms, because they have different depth and velocity preferences. The selection of an appropriate flow regime for a river requires clear goals and target objectives, with levels of protection set according to the relative values of the in- and out-of-stream resources. The process of establishing target objectives needs to be focused – objectives should be relevant, important, flow dependent and hierarchical. Failure to establish clear management goals and to carry out wide consultation will lead to conflict. Attempts to maintain everything in the existing state invariably lead to the conclusion that flows should not be changed, and precludes the opportunity for enhancement of some aspects of the aquatic environment and use of the water resource.

13. Acknowledgments

We would like to thank the many field staff of NIWA and Regional Councils who over the past 30 years have assisted in the collection of physical and biological data that have been used for the various analyses in this guide. In particular, Barry Biggs and Jody Richardson have been instrumental in collecting fundamental biological data that is necessary for instream habitat suitability criteria. We would also like to thank Joe Hay of Cawthron Institute for his careful review and the Foundation for Research, Science and Technology (New Zealand) for funding under contract C01X0308.

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