CAWTHRON

Instream Habitat Modelling and its Application in the Motueka Catchment

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Prepared for

INTEGRATED CATCHMENT MANAGEMENT for the Motueka River

•ridge tops to the sea•

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EXECUTIVE SUMMARY

Management of water allocation is a major issue throughout New Zealand and involves consideration of instream flow needs along with the potential benefits associated with out-of-stream water uses. Managers are faced with the question of how much water can be abstracted from a river without causing significant harm to the river ecosystem, or affecting the cultural, aesthetic and recreational values associated with that river system.

A range of methods are available to help guide decisions on flow management. This report provides an overview of the available methods and then compares the output from two methods (1-dimensional habitat modelling and 2-dimensional habitat modelling) in a short reach on the Motupiko River. Two-dimensional modelling is then applied to a reach on the Motueka River upstream of Tapawera to provide guidance on an appropriate minimum flow for this reach of the river.

Methods to help assess how much flow ought to be left in rivers can be divided into three general types: historic methods, hydraulic methods and habitat methods. Historic methods are the simplest and easiest to apply, but are probably most appropriate for river systems where the linkages between ecosystem integrity and flow requirements are poorly understood. Hydraulic methods require some field survey work and predict how depth, velocity and river width will change with flow. Habitat methods are the most sophisticated means to quantitatively assess the instream flow requirements of rivers and relate changes in depths and velocities at different flows with the habitat requirements of particular species. However, these methods require intensive field surveys and still attract controversy regarding the scientific defensibility of the habitat suitability criteria and model outputs. Some new methods involving models based on energetics concepts have significant potential in the near future, but also require intensive field surveys.

One-dimensional (1D) habitat methods are based on field surveys of river cross-sections, while two-dimensional (2D) habitat methods are based on detailed surveys of riverbed topography throughout specified reaches. Both methods require a rating curve to determine how the water surface elevation in at least part of the reach will change with flow. Within the range of the rating curve, 1D models are easier to calibrate than 2D models. However 2D models give a better measure of the longitudinal variation in depth and velocity than 1D models and probably predict changes in velocity distribution outside the calibration range more accurately than 1D models. 2D models are also able to predict how braiding patterns will change with flow in braided rivers.

A comparison of 1D and 2D habitat modelling was conducted in a short reach in the Motupiko River to demonstrate the processes involved and the type of outputs from each approach. The surveys involved were not as extensive as would be required for the actual application of these techniques, so the results should be interpreted with caution and not used as the basis of flow management decisions. The 2D approach has the advantage that the distribution of suitable habitat throughout a particular reach can be plotted in a plan view (map of the river reach) and the changes related to flow clearly shown. However, it is important to use an appropriate metric of habitat suitability when producing these plots. Due to differences in computational mesh density within the 2D model in different parts

of the reach, it is not appropriate to plot plan views of area weighted habitat suitability (WUA). Unweighted suitability scores should be used in such plots.

In general the response of predicted habitat suitability to flow for most species and life stages was reasonably comparable between the 1D and 2D approaches, although there were some differences in the flows that were predicted to result in peak habitat availability for juvenile brown trout using one set of habitat suitability criteria. Binary suitability criteria that simply distinguish between suitable and unsuitable habitat were also trialled. The advantage of using this binary approach is that the area weighted sum of its values actually represents an area of suitable habitat, rather than just a dimensionless index. However, in this reach and using the range of habitat suitability criteria that were applied, the binary approach essentially reduced the habitat method to an index of wetted area rather than an index of quality habitat. A binary threshold approach was also trialled where only habitat scoring above a certain threshold was considered suitable. The remaining habitat was considered unsuitable. The binary threshold approach provides an indication of how the area of optimum habitat changes with flow, however there is a degree of subjectivity in deciding what threshold level to use in distinguishing between suitable and unsuitable habitat. The other major issue with applying these binary systems is that the suitability criteria that are available have not been developed with this type of binary suitability in mind.

The 2D habitat modelling approach was applied to a 400 m reach in the Motueka River upstream of Tapawera with the aim of providing guidance on a minimum flow requirement for this section of the river. There are substantial losses of surface water to the aquifer in this reach of the river and thus this reach was considered to represent the area that will first experience the impacts of low flow. Flow management decisions need to consider the critical values of the reach of interest and provide flows capable of retaining a percentage of the habitat for the critical value that would be available at the mean annual low flow (MALF). The concept of critical values is based on the premise that if sufficient flow is provided to sustain the most flow sensitive, important value, then the other significant values will also be sustained. Using adult brown trout as the critical value in this reach of the river, a flow of 1.2 m^3s^{-1} is predicted to maintain 90% of the habitat available at the MALF. However, it is generally recognised that minimum flows must be set in conjunction with appropriate allocation rules to ensure that a degree of the natural flow variability is maintained. We suggest that a factor that could be considered in this process is to ensure that the invertebrate habitat at the median flow is not reduced excessively by water allocation. This would provide a biological rationale for the level of allocation in addition to that underpinning the setting of the minimum flow. Invertebrate habitat at the median flow is relevant to maintenance of the productivity of invertebrate populations, which provide the food base for fish.

All of these modelling efforts are based at a reach scale and therefore the information is really only applicable to the parts of the Motueka River Catchment where the field surveys and modelling have been conducted. Ideally, it would be useful to look at flow management in the catchment as a whole, rather than on a reach by reach basis. Over the next few years we will be attempting to scale-up some of this reach scale information to better understand how habitat availability changes with flow throughout the catchment.

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1. INTRODUCTION

A range of instream flow methods are available to support flow management decision making. These methods can generally be categorised into three major groups; historic, hydraulic and habitat methods (Jowett 1997). Historic methods base flow decisions on the historic flow regime. Hydraulic methods take account of the way various parameters describing the hydraulic geometry of the channel (often the wetted perimeter, or the wetted width) change with flow (Jowett 1997). Instream habitat modelling combines hydraulic modelling with species and lifestage specific habitat suitability criteria to predict how the availability of suitable physical habitat will respond to changes in flow.

Detailed one-dimensional habitat modelling has been conducted previously at two locations in the Motueka Catchment – Woodstock and Woodmans Bend (Hayes 2002) and also in a reach of the neighbouring Riwaka River (Hayes 1998). Simpler hydraulic and habitat methods have also been tested in the Rainy River (Young & Hayes 2002). Information from these modelling efforts has been useful for guiding decisions on water allocation in these parts of the catchment. However, flow management is also a big issue in other parts of the catchment and therefore it is timely to review and compare the range of instream flow methods that are available and could assist with water allocation management in the future.

This report provides an overview of the available instream flow methods with a focus on habitat modelling approaches. In particular, one-dimensional (1D) and two-dimensional (2D) approaches to habitat modelling are compared, through application of these methods to a reach in the Motupiko River. Two dimensional habitat modelling is then applied to provide a minimum flow recommendation for the Motueka River upstream of Tapawera.

2. REVIEW OF INSTREAM FLOW METHODS

In a recent report to Environment Southland, Jowett & Hayes (2004) provided a reasonably comprehensive review of instream flow methods. Most of the following review has been taken directly from that report (with the permission of the authors).

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 30 years, there is no universally accepted method for all rivers and streams, and there are very few case studies of ecological response to flow changes that can be used to judge the success or failure of different methods. 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 methods that consider the flow regime, with some degree of flow variability incorporated to maintain the natural morphology and ecosystem.

Instream flow methods can be conveniently divided into three types: historic flow, hydraulic, and habitat methods. The methods were described by Jowett (1997) and are summarised in the following sections.

2.1. 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 rules-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 MALF (mean annual low flow), 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 having the flow regime deviate widely 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 a 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 (Figure 1). 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.

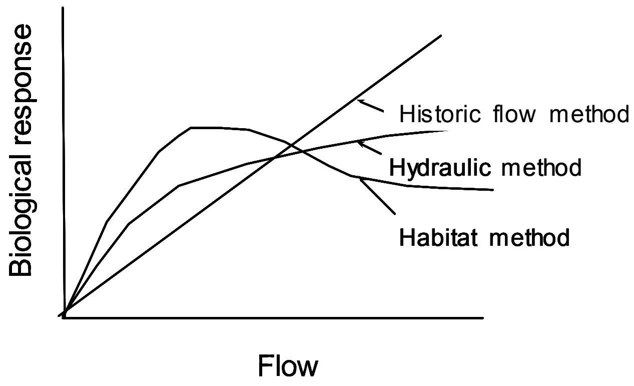


Figure 1. Hypothetical relationships between assumed biological response to flow for the historic flow, hydraulic and habitat methods. The biological response is assumed to be proportional to the flow, the wetted perimeter or width, and the weighted usable area, for the historic flow method, the hydraulic 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 11 United States of America streams (including streams in Montana) and an assessment of the depths and velocities needed for sustaining the aquatic life. Tennant found that at 10% of average flow the average depth was 0.3 m and velocity 0.25 ms^{-1}, and he 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 ms^{-1} and considered these to be in the good to optimum range for aquatic organisms. This is an example of a "regional method", applicable to a 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 Instream Flow Incremental Methodology (IFIM) predictions (e.g. Allan 1995; Hayes 2003). 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.

These are low risk approaches aimed at maintaining an ecosystem in its existing state and preclude the possibility that a river ecosystem could 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.

2.2. Hydraulic geometry and channel mapping methods

Hydraulic methods are more time consuming in that they are 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 much 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 (Figure 1) 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 depth and velocity requirements of some species.

2.3. Habitat methods

Habitat methods, including the habitat component of the IFIM, are an extension of the hydraulic methods. Their great strength is that they quantify the change in habitat availability and quality caused by changes in the natural flow regime, which helps the evaluation of alternative flow proposals. According to a review by the Environment Agency in the United Kingdom 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 in South Africa 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" (Tharme 1996). A review of flow assessment methods in the book "Instream flows for riverine resource stewardship" (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 development of the IFIM, in particular the hydraulic and habitat models (e.g. PHABSIM) (Mathur et al. 1985; Scott & Shirvell 1987; Kondolf et al. 2000; Hudson et al. 2003). A multi-authored review exposed divergent opinions regarding the scientific defensibility of PHABSIM (Castleberry et al. 1996).

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.

2.3.1. *Habitat models*

Several 1D computer models have been the mainstay for the evaluation of physical habitat, water temperature and sediment processes. Current software includes:

- PHABSIM (physical habitat simulation; Bovee 1982; Milhous et al. 1989).
- RHABSIM (river habitat simulation) used in the United States of America.
- RHYHABSIM (river hydraulic habitat simulation; Jowett 1989) used in New Zealand.
- EVHA (evaluation of habitat; Ginot 1998) in France.
- CASIMIR in Germany (Jorde 1997).
- RSS (river simulation system; Killingtviet & Harby 1994) in Norway.

Recently, 2D and 3D modelling software has been developed and used to predict flow patterns in complex rivers (e.g. River2D: www.river2d.ualberta.ca, and NIWA's 2D model – Beffa 1996; Duncan & Carter 1997; and SSIIM a 3D model: www.bygg.ntnu.no/~nilsol/ssiimwin). 2D models cope with braided channels, and complex cross flows (such as diagonal bars) better than 1D models. However, for most applications, on primarily single channel rivers, 1D models provide more accurate predictions and are more cost effective.

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. However 2D models do not necessarily predict water velocities accurately. Williams (2001) pointed out that velocity prediction was poor ($r^2 = 0.09$) in a 2D model of a 1,500 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 waters, 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 then 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.

In any model, the quality of the results will depend on the quality of the fieldwork 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 a 2D model (Duncan & Hicks 2001) and a 1D model (Mosley & Jowett 1985) were similar. In the Ashley River, Mosley & 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 14.4 m^3/s to 0.083 m^3/s. Duncan & Hicks (2001) compared measured and predicted depths and velocities in the Rangitata River and found 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.

Biological input to habitat models is supplied by 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. 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. The suitability curves in Figure 2 were developed for large, feeding adult brown trout in New Zealand (Hayes & Jowett 1994) and specify higher depth and velocities than curves for adult brown trout developed in the United States of America (Raleigh et al. 1986). This is likely to be due to the inclusion of resting fish locations in the development of the latter, while only actively feeding fish were included in the former. Differences in the sizes of fish may also contribute to the differences between these curves, but this 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.

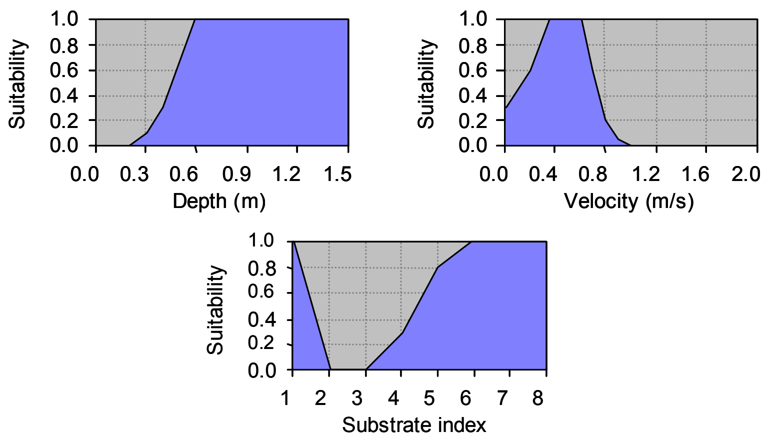


Figure 2. Habitat suitability curves for adult brown trout (adapted from Hayes & Jowett 1994).

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. 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).

The analysis can be separated into a hydraulic component and a habitat component. The hydraulic analysis predicts velocity and depth for 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.

The habitat analysis starts by choosing a particular species and life stage, and a particular flow. For each cell in the grid, velocity, depth, substrate, and possibly other parameters (e.g. cover) at the given flow are converted into suitability values, one for each parameter. These suitability values can then be combined (usually they are multiplied) and multiplied by the cell area to give an area of usable habitat (also called weighted usable area, WUA). Finally, all the usable habitat cell areas can be summed to give a total habitat area (total WUA) for the reach at the given flow. Although WUA is often interpreted as the area of usable habitat, it only represents an area when binary habitat suitability curves are used (i.e. habitat variables are either suitable (1) or unsuitable (0)). It is more correct to think of WUA as an area weighted index of available habitat.

This whole procedure is then repeated for other flows until eventually the outcome has been produced: a graph of weighted usable habitat area versus flow for the given species. This graph has a typical shape as shown in Figure 3 with a rising part, a maximum and a decline. The decline occurs when the velocity and/or depth exceed those preferred by the given species and life stage. In large rivers, the curve may actually predict that physical habitat will be at a maximum at less than naturally occurring flows. Thus, in contrast to the historic flow method, the habitat method does not automatically assume that the natural flow regime is optimal for all aquatic species in a river.

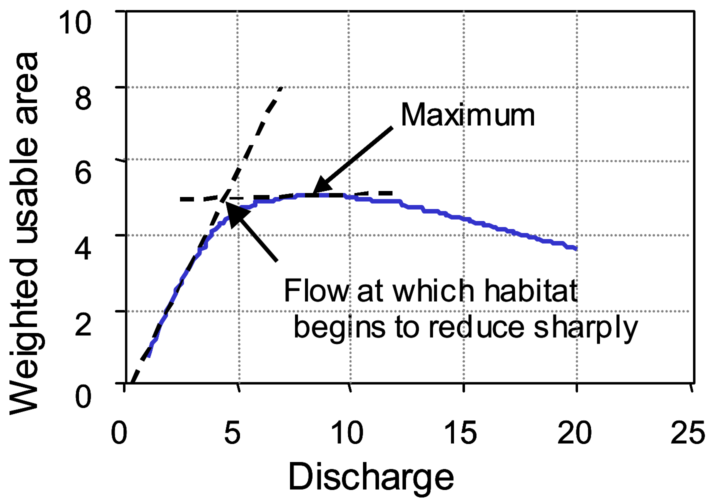


Figure 3. Selection of minimum flow at the breakpoint where habitat begins to decline sharply with decreasing flow.

The relationship between habitat and flow (Figure 3) 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 the mean annual low flow or median flow (or some other ecologically relevant flow statistic). It can also be defined as the flow that has the lowest acceptable minimum amount of habitat in absolute terms.

If the recommended minimum flow is at or above the habitat maximum 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 (Figure 3) and the difference between maximum habitat and the amount of habitat at a high flow is relatively small. 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 occurs at less than the minimum flow, it suggests that a reduction in flow might enhance those values.

Habitat suitability curves have been developed for threatened species (e.g. blue duck; Collier & Wakelin 1995), for species of special interest (especially trout and salmon; Hayes & Jowett 1994) and even for recreational activities (Mosley 1983). 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 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 may 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. If flow requirements of individual species are different, a solution may be found by choosing one with intermediate requirements (Jowett & Richardson 1995) or to define flow requirements for aquatic communities.

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 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 1999) (See Section 3.6 below).

2.3.2. *Fish models*

The most recent modelling advances have been on models that predict relationships between flow and fish themselves, rather than a habitat index (e.g. WUA). These ‘fish’ models have been developed for salmonids and some are at a stage where they can be used for flow assessment. These 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; Hayes et al. 2000, 2003; Guensch et al. 2001). 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 mainly to representative reaches at the scale of individual riffle/pool or run/pool units and are more expensive to run than traditional habitat modelling methods. Consequently they have a narrower range of applications. They are most appropriate when instream resource values are high (e.g. highly valued salmonid fishery) and stakeholders agree that fine scale modelling would be informative. They also have potential to complement conventional WUA based modelling (1D and 2D) (undertaken at broader spatial scales) by verifying whether predictions from the latter can be trusted. To date these models have been used only in research contexts in New Zealand but they have considerable potential in the applied arena.

Fish models operate on the output of hydraulic models and incorporate habitat features and foraging behaviours. Drift foraging models used in fish models provide a functional understanding of drift feeding and velocity use (Hughes & Dill 1990; Addley 1993, 2006; Hill & Grossman 1993; Hughes et al. 2003). The most advanced fish models link hydraulic models with invertebrate drift dispersion and drift foraging models to predict net rate of energy intake and growth potential (either for the average individual at the reach scale, or spatially explicitly), and carrying capacity (Hayes et al. 2000, 2003; Kelly et al. 2005).

2.4. 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 the proportion can vary 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 MALF 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, as with habitat methods, 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. However, regional methods are necessarily coarser in resolution than habitat methods, being in essence 'rules of thumb' and consequently do not provide the same level of detail, as full habitat methods, on how habitat varies with flow in specific rivers.

The rationale for habitat based regional methods is the same as 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 ms^{-1} might be considered poor, 0.1-0.3 ms^{-1} adequate, and 0.3-0.5 ms^{-1} 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 and are likely to result in flow assessments that provide life sustaining flows whilst retaining some degree of the river's character. In terms of the information that they can provide to flow managers they fall between relatively simple to apply historical methods and more complex and data intensive habitat methods.

2.5. Generalised instream habitat models

Habitat methods and instream habitat models have been used for many minimum flow studies in the last two decades (Gore & Nestler 1988; Reiser et al. 1989; Gallagher & Gard 1999; Guay et al. 2000). As described above, conventional instream habitat models link a traditional hydraulic engineering model 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 the 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 value (HV, ranging between 0 and 1; called HSI in RHYHABSIM or %WUA in earlier versions), and when weighted by surface area and summed over the reach, HV gives the weighted usable area (WUA). Therefore, the major reach-scale outputs of these models are relationships between WUA and discharge.

Applying conventional instream habitat models in a stream reach requires considerable field effort and experience. It involves a complete survey of bed topography and precise measurements of current velocities and water depths along several geo-referenced cross-sections, depending on the form of hydraulic model. The hydraulic model also requires calibration at two or more flows.

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 conventional instream habitat models 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 and field effort, and the models provide HV and WUA curves that 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 a range of taxa were predictable from simplified hydraulic data. Reach hydraulic geometry (mean depth and mean width-discharge relationships), average bed particle size and mean natural annual discharge 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 France. The Reynolds number of reaches (discharge per unit width) governs changes (pattern or shape) in habitat value for each species within reaches. The Froude number at the mean natural discharge, which indicates the proportion of riffles in stream reaches, was generally the major variable governing overall magnitude of habitat value in the different reaches. This is consistent with the preference of benthic fauna, such as many of the native New Zealand fish species and benthic invertebrates, for riffles (Jowett & Richardson 1995; Jowett 2000), and 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). This suggests that the generalised model equations can be used to model habitat quality anywhere in the world for taxa with comparable microhabitat suitability, at least within their calibration range. Generalised models necessarily lose some information compared to conventional models such as PHABSIM or RHYHABSIM. This loss must be balanced against the requirement for fieldwork and experience in conventional modelling. In particular, hydraulic geometry relationships in reaches 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. Therefore, detailed topographies of stream reaches, associated velocity measurements and hydraulic model calibration are not required.

The generalised habitat models described in Lamouroux & Jowett (2005) take the form:

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

Where the values c and k describe the shape of the relationship between the dimensionless habitat value (HV) and discharge (Q , m^3s^{-1}) per unit width (W, m), and 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.

HV is equivalent to expressing weighted usable area as the proportion of river width and it can be converted to the equivalent of WUA (in m^2/m) by multiplying by the river width at each flow.

These generalised models can be implemented in WAIORA.

2.6. WAIORA – implements water quality and generalised habitat models

2.6.1. *Water quality models*

WAIORA (Water Allocation Impacts on River Attributes) is a decision support system, developed by NIWA, with particular utility in cases where changes to a flow regime are considered likely to impact on water quality. It 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. 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. The generalised models described in the previous section can be implemented in WAIORA.

A number of assumptions have been made during model development (these are detailed in a manual and help file that can be downloaded from www.niwa.co.nz/ncwr/tools/waiora) and the outputs reflect the nature of these assumptions, as well as 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.

The quality and scope of the instream habitat survey data will determine the reliability of the results, particularly the degree to which you can extrapolate beyond the flows that were surveyed. Two levels of survey are available. For quick assessments, stream widths and depths can be measured at two flows in at least three locations in each habitat type (e.g. pool, run, and riffle). Stream width, depth and velocity are then estimated assuming logarithmic hydraulic relationships (Jowett 1998). In cases where you want to extrapolate to flows higher or lower than those surveyed, cross-section data can be collected and calibrated in RHYHABSIM. The normal procedure is to survey at least five cross-sections in each mesohabitat type (e.g. pool, run, and riffle) and re-measure water levels at two or more flows.

Calibration data can also be collected for water temperature and dissolved oxygen models. These calibration data should be collected at times of maximum stress, normally mid-summer. DataSondes can be deployed to measure diurnal variation in water temperature and dissolved oxygen concentration and inexpensive temperature loggers are available. Water temperatures are required at both the start and end of the section of river for calibration of the water temperature model. Although it is possible to model water temperature and dissolved oxygen without calibrated models, calibration is desirable to calculate appropriate parameters and coefficients for the dissolved oxygen models and to set appropriate initial water temperatures for the water temperature model.

Once the models have been calibrated, WAIORA calculates how stream width, depth and velocity, water temperature, and dissolved oxygen and ammonia concentrations vary with flow and displays the values of these parameters for the current low flow and the low flow that will result from the proposed abstraction or flow discharge.

2.6.2. *Generalised habitat models*

Stream width at flow predictions made by WAIORA can be used to calculate discharge per unit width (Q/W) for a range of flows which provides the input for predicting HV (habitat value) in generalised habitat models.

3. ECOLOGICALLY RELEVANT FLOW STATISTICS FOR MINIMUM FLOW SETTING

When setting minimum flows for instream values the assumption is made that low flow is a limiting factor. Research in New Zealand indicates that the mean annual low flow and median flows are ecologically relevant flow statistics for trout carrying capacity and stream productivity. Jowett (1990, 1992) found that instream habitat for adult brown trout at the mean annual low flow (MALF) was correlated with adult brown trout abundance in New Zealand rivers. The habitat metric that he used to quantify instream habitat was percent WUA (equivalent to HSI). The adult brown trout habitat suitability criteria used in Jowett's analysis were developed by Hayes & Jowett (1994). The inference arising from Jowett's research was that adult trout habitat (WUA%) at the MALF acts as a bottleneck to brown trout numbers. He also found that invertebrate food producing habitat (WUA%, defined by Waters' (1976) general invertebrate habitat suitability criteria) at the median flow was strongly associated with trout abundance (Jowett 1990, 1992). These two habitat metrics are surrogate measures of space and food, which are considered to be primary factors regulating stream salmonid populations (Chapman 1966).

The MALF is indicative of the low flows likely to be experienced during the generation cycles of trout. Brown trout usually mature at between two and five years of age, with age three for first spawning being most common in rivers. On average a trout makes the greatest reproductive contribution to the population over the first two or three years of spawning. The MALF has an expected return period of about 2.33 years in most rivers. Consequently, the MALF sets the lower limit to physical space likely to be experienced by trout before they are able to begin making a reproductive contribution to the population (i.e. it may be a factor in limiting the number of trout that are able to be supported through to reproductive age).

Long-term (30 years) research by J.M. Elliott, and others, of a sea-run brown trout population in a small English stream (Black Brows Beck) has shown that, droughts with a 10 to 30 year return period, and even those with less than a five year return period, can have significant impacts on the reproductive capacity of a trout population (Elliott et al. 1997; Bell et al. 2000).

The effects of sustained drought were to reduce the stream area available as habitat and retard trout growth, both factors contributing to high mortality of juvenile trout (0+ and 1+) [Adult trout were not resident in this stream] (Elliott 1994; Elliott et al. 1997). Reductions in growth and abundance of juvenile trout (0+ and 1+ trout) due to drought translated to reduced numbers of returning adults and consequent egg production.

Droughts in two successive summers had a compounding impact on survival, reducing one year class in their first year to 40% of their expected numbers, and to 5% of expected numbers in their second year (Bell et al. 2000). Based on a stock recruitment model, repeated droughts (e.g. in successive years) were predicted to cause cyclic or even chaotic behaviour in the trout population (Bell et al. 2000). Constant drought conditions were predicted to reduce the population growth rate, abundance, and variability in abundance from year to year; these effects being dependent on the severity of the drought (i.e. rainfall or flow reduction). Severe constant drought conditions, equivalent to about a 1-in-10 year drought or worse, were predicted to result in a declining population (Bell et al. 2000). We can infer from this prediction that if abstraction were to routinely draw flows down to the equivalent of the natural 1-in-10 year low flow in small rivers then the trout population would go into decline.

This research was on a sea run brown trout population. It might be expected that sea run trout would cope better with low flow events than resident populations because they can escape the effects of low flow on the adult life stage. Sea-running is considered, in part, to be a response to unfavourable habitat conditions in the freshwater environment (Thorpe 1994). Nevertheless, despite being sea-run, the Black Brows Beck trout population suffered significant impacts from droughts with 1-in-3 to 1-in-10 year return periods. There were significant reductions in survival and growth of juvenile trout resulting in fewer returning adults, and egg production of affected year classes was reduced by 73–83% (Elliott et al. 1997). It is reasonable to expect that the effects on resident trout populations in similar small streams would be even greater.

National and regional differences in climate and hydrology make a direct transfer of Elliott's results from Black Brows Beck to New Zealand rivers uncertain. Nevertheless, the response of trout in Black Brows Beck to droughts highlights the vulnerability of small stream trout populations to low flow events. In this stream, droughts, and probably low flows, equivalent to, and less than, the seven day 1-in-10 year low flow had significant impacts on the trout population. Impacts of droughts of similar return frequency might be expected to be less severe in larger rivers, and differences in channel morphology are likely to affect the sensitivity of instream habitat to flow reduction. So too will catchment geology through its affect on the rate of flow recession and base flow (Duncan 1992).

The MALF is closely correlated with annual low flow events, and as such also provides an index of the minimum flow that can be expected from year to year (although the one year return period minimum flow would arguably be a more relevant statistic). The lowest flow that a river falls to each year sets the lower limit to physical space available for adult trout, although the duration of low flow is also relevant. This annual limit to living space potentially sets a limit to the average numbers of trout. This concept is intuitively sensible to anyone who has spent a lot of time looking for trout in rivers. Rivers that fall to very low flows each year hold few trout while those that sustain high low flows hold a lot of trout.

It seems reasonable that the MALF should be similarly relevant to native fish species with generation cycles longer than one year, at least in situations where habitat declines toward the MALF. If the minimum flow restricts habitat for any species, there is potential for a detrimental effect on that population. NIWA research in the Waipara River, where habitat is limited at low flow, showed that the detrimental effect on fish numbers increased with the magnitude and duration of low flow. An instream habitat survey (Jowett 1994) showed that fish habitat began to decline sharply when flows fell below 120 L/s, slightly greater than the 7-day MALF of 112 L/s. In the first summer (1998/99 mean flow 1190 L/s), daily mean flows were less than 120 L/s for 31% of the time and fell to 32 L/s. In this year, there was a substantial decline in abundance of three of the four common native fish species in the river. The following summer (1999/00 mean flow 1243 L/s) there was little change in native fish abundance when daily mean flows were less than 120 L/s for 10% of the time and fell to 69 L/s. In the third year, flows were less than 120 L/s for 61% of the time and fell to 47 L/s, and two of the four common fish species declined in abundance (Jowett & Hayes 2004). Research on the Onekaka River, in Golden Bay, also showed that when habitat availability (estimated by WUA) was altered by flow reduction, abundance of three native fish species showed responses similar to those in habitat availability in both direction and magnitude (Richardson & Jowett 2006) (i.e. eels and koaro habitat was reduced and these species declined in abundance, while redfin bully habitat increased and so did their numbers).

By contrast to long-lived species such as trout, most benthic invertebrates have generation cycles less than or equal to one year, and they can be repopulated from tributaries and from other rivers by winged dispersal. Recolonisation of some river beds by benthic invertebrates following floods has been reported to occur within 4-6 weeks. In other words, the abundance of benthic invertebrates can respond relatively quickly to available habitat conditions, so their populations respond to more frequent limiting events (e.g. floods or low flows that occur over the time-scale of months). The median flow provides an approximation of the habitat conditions experienced, and able to be utilised, by benthic invertebrates most of the time.

The above rationale provides the conceptual ecological basis for interpreting trout WUA x flow curves with respect to the MALF. If protecting habitat for fish, with longer than annual life cycles, is a factor in setting a minimum flow condition then the MALF is an ecologically defensible choice for a conservative minimum flow. However, this may leave little or no flow available for existing out-of-stream users in most years. If a low flow condition lower than the MALF is to be considered to allow for both instream and out-of-stream flow requirements then the WUA x flow curves can be used to determine percentage habitat reduction with incremental flow reduction and so serve as a basis for minimum flow negotiation between the various interested parties. The results may also be referenced to other historical flow statistics (e.g. the 1-in-5 year low flow). Maintenance of invertebrate production is arguably more dependent on allocation limits or flow sharing rules, which ensure that the median flow is not substantially reduced by abstraction, than on the minimum flow per se.

4. A COMPARISON OF 1D VERSUS 2D HABITAT MODELLING ON THE MOTUPIKO

To provide a comparison between the 1D and 2D approaches to habitat modelling, and the types of outputs that each produce, we applied both techniques to a short reach in the Motupiko River. It must be emphasised that the 1D modelling described here was undertaken only to provide this comparison. Due to budget constraints, the survey involved was not as extensive as would be required for an actual application of this modelling technique to support flow decision making. Consequently, the results should be interpreted as indicative only, and should not be used as the basis for flow management decisions. The 2D survey was also restricted to a relatively short reach (a single run riffle pool sequence). However, given that the 2D approach provided a relatively detailed description of physical habitat through the modelled reach, the results may be applicable for flow setting (assuming that the modelled reach was reasonably representative of the section of river that the flow decision would affect).

4.1. Methods

4.1.1. Field surveys

A 120 m long reach on the Motupiko River containing a single riffle, run, pool sequence was surveyed on 8 December 2004. The flow during this survey was 1.8 m^3s^{-1}. Subsequent additional data were collected from the reach on 9 February 2005, at a flow of 0.6 m^3s^{-1}.

The field survey involved making a reasonably detailed topographical survey of the streambed through the reach using a total station (Trimble 5600-series). The aim during this survey was to obtain the best possible description of the bed topography using as few data points as necessary. This was achieved by concentrating on surveying breaklines (lines marking notable changes in slope e.g. the top and toe of stream banks, the channel thalweg, heads and tails of riffles) to get an overall description of the channel form, then surveying additional points to provide a more detailed description of the variation in topography between these breaklines.

This methodology resulted in a data set consisting of 445 points to describe the streambed topography of this reach. However the reach surveyed in this instance was short and had a relatively simple channel form. More extensive reaches or ones involving more detailed surveys (e.g. multiple run, riffle, pool sequences, or complex braided channels) may be expected to require a week, or more, of field work to provide an adequate topographic description (Steffler & Blackburn 2002).

As well as the topographical survey, flow was gauged at the top of the reach and depth and velocity measurements were made across three other cross-sections spaced through the reach, including one at the end of the reach. The measurements from these cross-sections were later used to calibrate the 2D flow model. The cross-sections were located so that they sampled the different micro-habitat types that occurred in the reach (i.e. run, riffle, pool). Stage measurements were taken at these cross-sections and again at the subsequent follow up survey, and these measurements formed the basis of a rudimentary 1D model of the reach (see Section 4.1.3) for comparison with the 2D model results.

4.1.2. The 2D hydraulic modelling process

The 2D flow and habitat modelling was undertaken using River2D (Steffler et al. 2003). This 2D hydrodynamic modelling software package was developed at the University of Alberta and is available free to download from their website. Habitat modelling in River2D is a multi-step process involving three linked pieces of software.

In the first stage the topographical survey data are plotted as a triangulated irregular network (TIN). Coloured contours can then be used to identify possible erroneous data points or unrealistic spikes in the bed form. These can be edited or corrected interactively (by defining additional breaklines, for example) until what is deemed to be a realistic representation of the bed form of the channel is achieved.

Bed roughness can also been defined interactively at this stage. This parameter contributes to control of the frictional effects of the bed on the modelled flow. Bed roughness was specified based on a series of polygons, which defined regions identified during the field survey as having relatively homogeneous dominant substrate size. The bed roughness parameter can be altered later as part of the calibration process, to improve the match between observed and modelled depths, velocities and water surface levels. Steffler & Blackburn (2002) suggest a value of 1-3 times the largest grain diameter in each region as a starting estimate.

The next stage involves developing a computational mesh (i.e. a series of computational nodes connected in an irregular network) that will ultimately form the framework for computation of the flow solution. This computational mesh interpolates the streambed topography from the original bed file. So in constructing the computational mesh it is important to ensure that an adequate description of the bed topography is reproduced. It is also important to consider the spacing of computational nodes so that complex flow features (e.g. eddies) will be adequately described in the final flow solution. These considerations need to be weighed against the computation time required by models with large numbers of nodes. The irregular network used in River2D has the advantage that nodes can be closely spaced in areas of complex flow, or topography, and be spaced further apart in areas with more simple flow or topography. The boundary flow conditions (i.e. the inflow discharge at the top of the reach, and the stage height at the bottom of the reach) are also specified at this stage. The computational mesh is then exported in the format required for input to the flow-modelling software component.

River2D then models the flow of water through the computational mesh, given the bed topography and the boundary conditions. It performs iterative calculations to attempt to solve the flow through the modelled reach, conserving mass and momentum of water between computational nodes. A flow solution is considered to have converged (i.e. balanced), when the change between successive iterations becomes very small and the inflow and outflow balance (within an acceptable margin of error). Flow solutions described in this report were taken to be converged when the solution change fell below 0.00001, and the modelled inflow and outflow balanced. However, there was always some degree of discrepancy in level of balance between the inflow and outflow, and this was generally greatest when modelling very high or very low flows relative to the measured flow that the model had been calibrated against.

The flow solutions for the measured flows were calibrated against the observed depth and velocity measurements taken across the cross-sections through the reach, and the modelled wetted perimeter was compared visually with the shape and location of the observed water's edge. Bed roughness values were manipulated (based on the areas identified during the field surveys as having relatively homogeneous substrate size) and the flow model rerun in an attempt to minimise the sum of the squared differences between modelled and observed depths and velocities across the calibration cross-sections. The bed roughness distribution that was developed through this process was then used as a base for modelling flows other than the calibration flow.

4.1.3. The 1D hydraulic modelling process

The 1D modelling was undertaken using RHYHABSIM (Jowett 2004) based on the water stage and discharge measurements made across the four cross-sections in the survey reach, at two flows. These stage height and discharge measurements, along with estimates of the stage at zero flow, were used to calculate stage-discharge relationships for each cross-section. The two flows measured at each cross-section, plus an estimate of the stage at zero flow, represent the minimum data required to develop these relationships. Usually three or more measurements would be used to develop more robust relationships between stage and discharge at each cross-section.

Also the small number of cross-sections (four over the surveyed reach) is substantially less than would normally be required in a 1D survey. Jowett (2004) recommends at least five cross-sections located in each habitat type, and a recent sensitivity analysis showed that 18-20 cross-sections were generally required to produce a robust relationship between predicted habitat availability and flow (Payne et al. 2004). However, these measurements do serve to provide a comparison between the 1D and 2D modelling processes and the type of results generated by these methods.

The substrate composition at the measurement points across the cross-sections were estimated from the map of dominant substrate types used to define bed roughness in the 2D modelling process.

The pattern of velocity distribution across the cross-sections was calculated automatically in RHYHABSIM from the gauging data at the survey flow. However, the velocity distribution factors for points that were above water level during the survey were edited to facilitate modelling of flows above the measured flow. The velocity distribution factors at these points were adjusted to conform to the expectation that they should decrease toward the banks (Jowett 2004).

RHYHABSIM predicts the distribution of water depth and velocity across the surveyed cross-sections at different flows. These cross-sections are assumed to be representative of the variation in habitat types in the rest of the reach, not covered by the cross-sections (although, given the small number of cross-sections in this example, the full range of variability of habitat through the reach was probably not represented in this case). The changes in predicted depths and velocities with flow at the cross-sections are therefore assumed to represent these changes throughout the habitat type that the cross-sections represent. The influence that each cross-section had on the model predictions was weighted by the percentage of the overall reach that each cross-section represented in terms of micro-habitat (i.e. pool, run, riffle).

4.1.4. *Habitat modelling*

The habitat modelling component of both the 1D and 2D modelling approaches is essentially the same. However, in the 2D approach an additional file giving a substrate size value for each node is also required, since the bed roughness parameter specified at each node is meant to represent the frictional forces of the substrate acting on the water flowing over the bed and may not represent the actual size of the substrate.

Suitability criteria that describe the physical habitat (depth, velocity and substrate) preferences of a given species (or life stage of a species) are compared with the predicted values for each of these physical habitat variables at each node, or point on each cross-section. The suitability of each physical habitat variable for the species in question is rated on an index from 0 to 1 (1 being ideal and 0 being unsuitable). These values are then usually combined, by multiplying them together, to give an overall suitability rating of the physical habitat at each node or point.

These values can then be summarised over the entire reach to provide a single statistic that can be compared between different flows. The traditional summary statistics used in RHYHABSIM are weighted usable area (WUA) and habitat suitability index (HSI, previously known as WUA%). WUA is calculated by weighting the combined suitability score at each point by the area of streambed represented by that point (by multiplying the suitability score by the area), then summing across all the points. This gives a dimensionless index of the relative quantity of suitable habitat predicted.

Although it is useful as a summary statistic, WUA does not lend itself well to the 2D plan view format of data presentation provided by River2D (e.g. Figure 4). This is because the area weighting gives the appearance of having devalued the suitability of nodes representing small areas of streambed, while overvaluing nodes that represent larger areas. These apparent biases are corrected in the summing process producing the single summary statistic, because the many low WUA values in areas with tightly spaced nodes will be equivalent to the few high values from areas where the nodes are sparsely distributed (e.g. 20 nodes representing 0.1 m^2 each, each having a combined suitability value of 0.8 will contribute a combined WUA of 0.1 x 0.8 x 20 = 1.6, c.f. four nodes each representing 0.5 m^2, with a combined suitability value of 0.8 would give 0.5 x 0.8 x 4 = 1.6, in both cases the overall area represented is 2 m^2 and the combined suitability value at each point in that area is 0.8).

When viewing the data as a colour contour map, the combined suitability values (unweighted) at each node give a better representation of the spatial distribution of suitable habitat predicted through the modelled reach (Figure 4). However, simply summing the combined suitability values at each node would bias the result toward the values of numerous tightly spaced nodes, each representing only a small area of streambed. This means that while the combined suitability values are well suited to 2D colour contour map views of the modelled reach, showing the predicted spatial distribution of suitable habitat, WUA provides a better summary statistic over the entire reach area. This is not generally an issue in traditional 1D habitat modelling because the summary statistic is the only output generated (i.e. there is usually no attempt to plot plan views of habitat suitability through the modelled reach). RHYHABSIM does provide a 2D plotting option, provided that the cross-sections have been surveyed in a way that makes this feasible. However, it plots the combined suitability indices in this format (i.e. depth x velocity x substrate suitability, without area weighting) not WUA (I. Jowett, NIWA, pers. comm.).

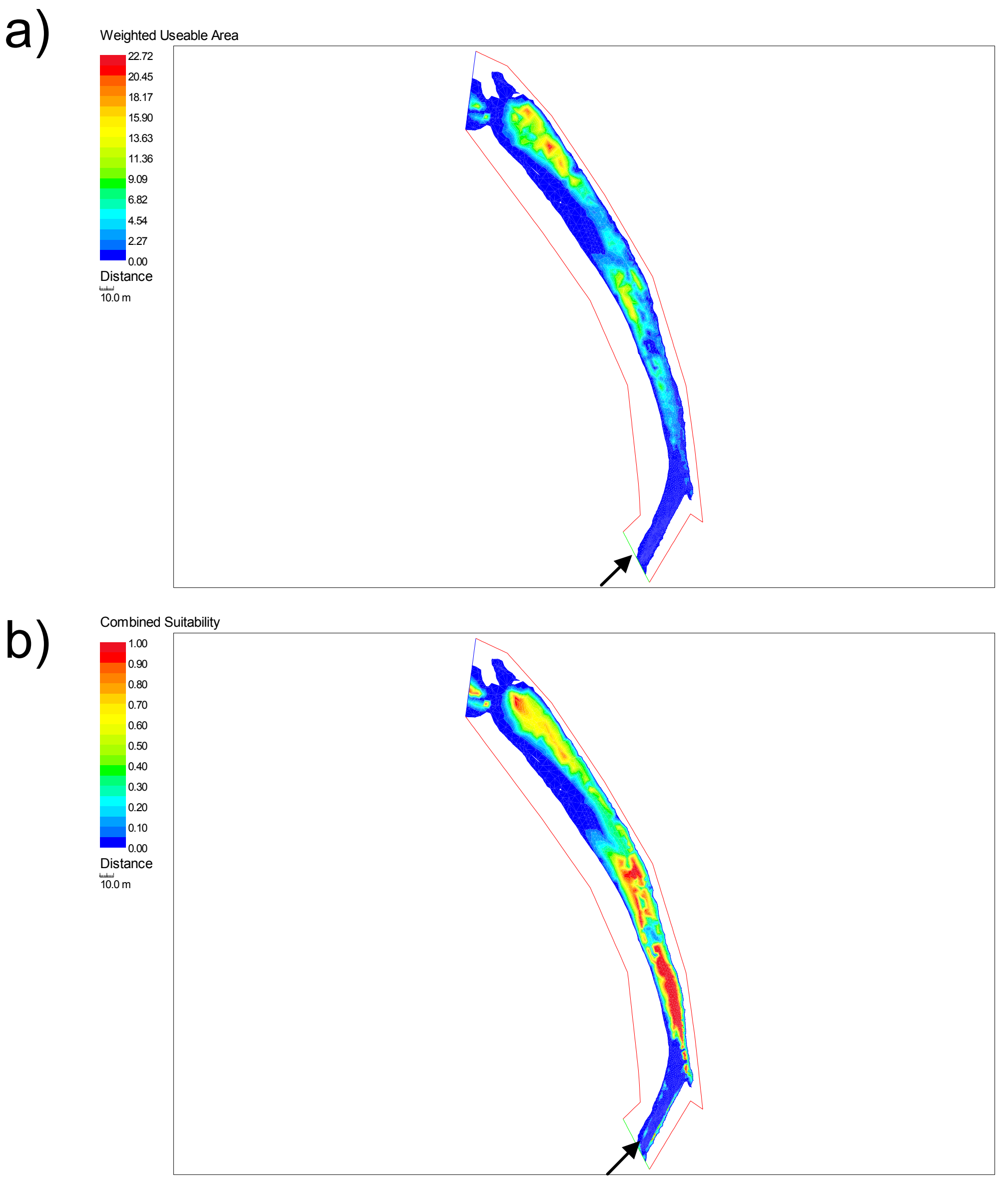


Figure 4. A comparison of 2D plan view plots of (a) WUA and (b) unweighted combined suitability scores predicted for adult brown trout at a flow of 4 m^3s^{-1} in the Tapawera modelled reach. Arrow indicates direction of flow.

The HSI (in RHYHABSIM) is calculated as the average of the combined suitability scores across all the points, and provides an index of the average quality of habitat predicted for the reach. This index is not calculated in River2D, but it can be calculated by averaging the combined suitability indices across all the nodes in a text output. However, it is also susceptible to bias toward the values of nodes representing small cells. HSI can range in value between zero and one, with one being ideal habitat throughout the reach (i.e. ideal at all surveyed points). In older versions of RHYHABSIM this index was multiplied by 100 and was referred to as WUA%.

An alternative method of treating the suitability values is to consider any habitat that is at all suitable as being "good", giving it a value of 1, and giving all unsuitable habitat a value of 0. This type of binary suitability system has the advantage that the area weighted sum of its values actually represents an area of suitable habitat, rather than being a dimensionless index. However, there are some disadvantages with this system. One is that it fails to distinguish between optimal, suboptimal and barely tolerable conditions (Bovee 1982).

A "binary threshold"criteria has been suggested as a way to circumvent this issue. Hudson et al. (2005) applied a threshold of 0.8 for each physical habitat variable (i.e. points with habitat suitability scores above 0.8 for each physical habitat variable are considered good and given a value of 1, all other points get a value of 0, "unsuitable"). This technique only considers very good habitat as being suitable;(under this system points with a combined suitability index ≥ 0.512 (i.e. $\geq 0.8^3$) are considered suitable) and treats all other more marginal habitat as unsuitable, but still has the advantage of providing a summary statistic that represents an area of suitable habitat (i.e. m^2). As such, this approach arguably provides a readily understandable metric of the predicted availability of very good physical habitat. However, it has the disadvantage of ignoring all habitat of lesser value and is sensitive to the arbitrary "threshold" level chosen.

The other major issue with applying either of these binary systems is that the suitability criteria available have generally not been developed with this type of binary suitability in mind (Denslinger et al. 1998). These criteria would arguably have been developed differently if they were intended to be used in a binary system of suitability (i.e. they would be binary rather than continuous from 0 to 1), which would require a different approach to their development (possibly logistic regression).

In this report we present all of these approaches to summarising the output of the 2D modelling and we present the traditional WUA and HSI statistics from the 1D modelling for comparison.

The flow range modelled (0.1 to 5 m^3s^{-1}) was selected to span the low flow range for the modelled reach, with an estimated MALF of 0.50-0.53 m^3s^{-1} (Martin Doyle, TDC, pers. comm.), but also to give an indication of the likely response of habitat availability to higher flows.

The species for which habitat availability was modelled were identified from fisheries distribution maps, based on the New Zealand Freshwater Fisheries Database (NZFFD), outlined in the Motueka integrated catchment management baseline information report (Basher 2003). Suitability criteria for two different size classes of longfin eels were included, and for brown trout four sets of suitability criteria were applied to cover different life history stages (Appendix 1). The other species included were dwarf galaxias, torrentfish and upland bullies (Appendix 1).

4.2. Results and discussion

4.2.1. WUA and HSI

Inflow and outflow in all but three flow solutions (0.1 m^3s^{-1}, 0.25 m^3s^{-1} and 5.0 m^3s^{-1}) balanced to within 2%, and only the 0.1 m^3s^{-1} solution was more than 3% out of balance (6% low). Consequently, the habitat availability predictions at 0.1 m^3s^{-1} are likely to be underestimated. Although in terms of the absolute difference between inflow and outflow, this imbalance equates to only six litres per second.

For most species and life stages the response of predicted habitat to flow in terms of HSI was reasonably comparable between the 1D and 2D methods (Figures 5 and 6). This means that the average quality of habitat predicted by both methods was similar across the modelled flow range. The notable exception to this was yearling – small adult brown trout feeding habitat predicted using Roussel et al.'s (1999) suitability criteria (Figure 5a). The HSI curves based on these criteria peaked at slightly different flows (approximately 0.5 m^3s^{-1} for the 2D versus 0.75 m^3s^{-1} for the 1D prediction). There was also a substantial difference in the magnitude of these peaks in terms of HSI. The 2D modelling predicted lower average habitat suitability than did the 1D modelling. This was possibly because the 1D version of the HSI was calculated based on relatively few data (a total of 76 points on all cross-section c.f. 3,912 computational nodes in the 2D model) and so would be more susceptible to being skewed by extreme values.

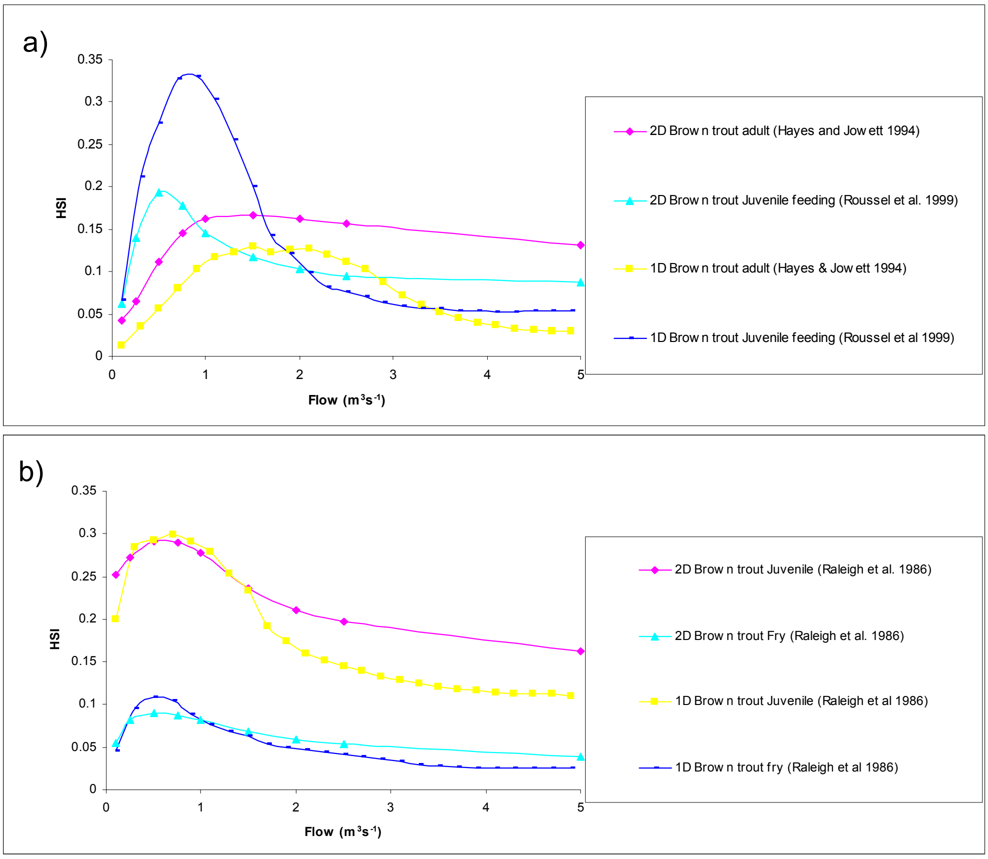


Figure 5. Comparison of HSI predictions based on 1D and 2D modelling for a range of brown trout life stages in the Motupiko modelled reach over a range of flows. (a) Adult and juvenile feeding habitat, (b) juvenile and fry habitat

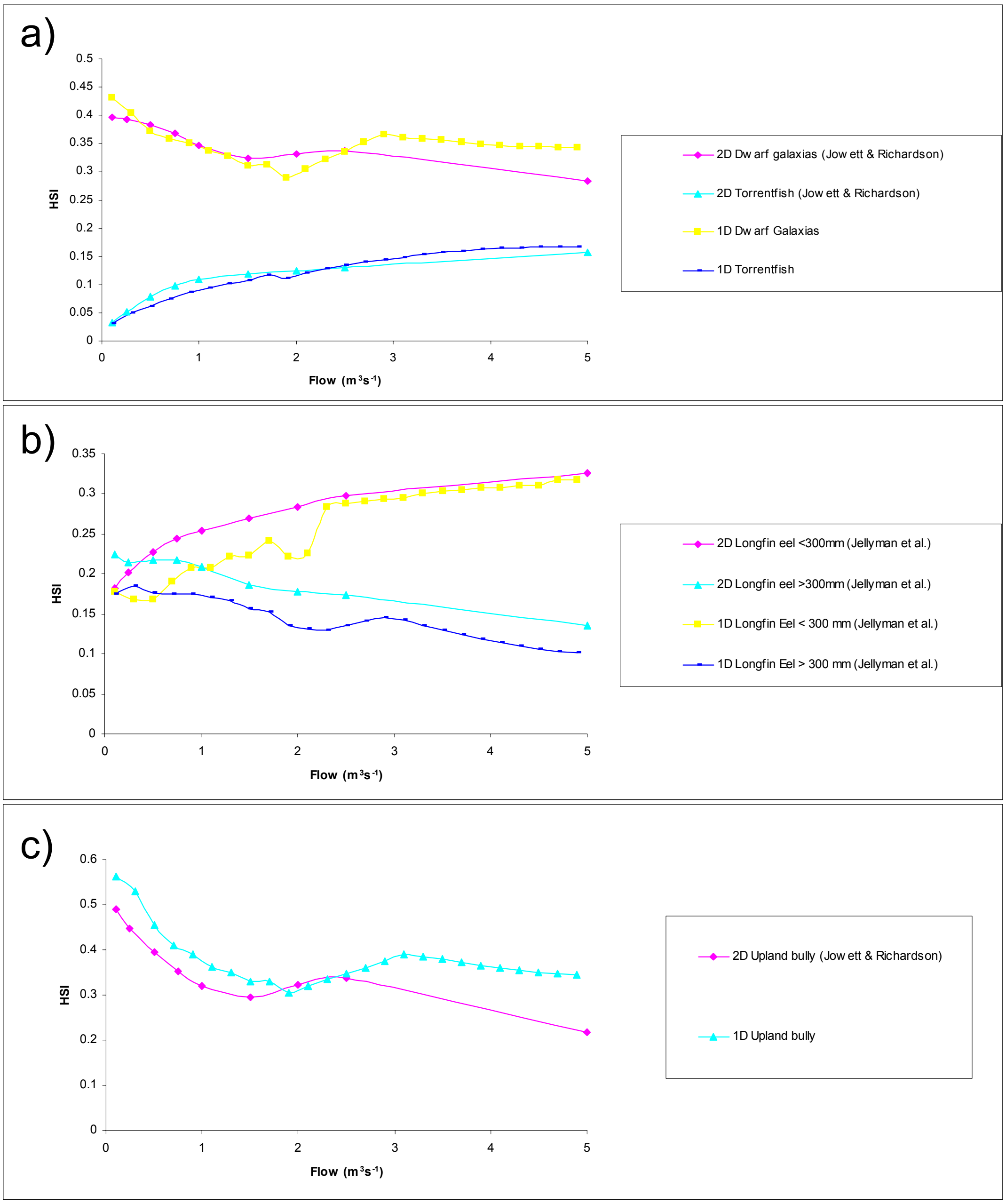


Figure 6. Comparison of HSI predictions based on 1D and 2D modelling for a range of native fish species in the Motupiko modelled reach over a range of flows. (a) Dwarf galaxias and torrentfish, (b) Longfin eel <300 mm and >300 mm, (c) upland bully

The very peaked nature of the HSI response curve based on Roussel et al.'s (1999) yearling – small adult brown trout criteria (Figure 5a) is probably due to these criteria having been developed in a stream with very low discharge. These criteria were developed in a small French stream with a discharge of approximately 0.11 m^3s^{-1}. While this arguably makes these criteria well suited to assessing the habitat availability for brown trout at low flows in this reach of the Motupiko, it does mean that as velocity and depth increase at higher modelled flows these criteria are likely to underestimate habitat availability. Since such extremes of depth and velocity may not have been available in the small stream where these criteria were developed, it may not have been feasible to assess the suitability of such depths and velocities as habitat.

The HSI response curve for adult brown trout, using Hayes & Jowett's (1994) suitability criteria, based on the 2D model also peaked at lower flows than the response curve based on the 1D model (Figure 5a). However in both cases the response curve had a fairly broad peak, with little change in the level of the HSI predicted over a range of flows, and importantly the break points i.e. points of rapid change in slope, occurred at similar flows.

At first glance, the WUA response curves appear to show greater dissimilarity between 1D and 2D modelling approaches than did HSI (Figures 7 and 8). However, this apparent dissimilarity is mainly due to a disparity in the magnitude of WUA achieved by the response curves based on the two modelling approaches. The shapes of the response curves were generally broadly similar for a given species (Figures 7 and 8). This is an important distinction to make because it is the shape of the curve, rather than its magnitude in terms of WUA, that is important when interpreting WUA versus flow curves to inform flow decision making.

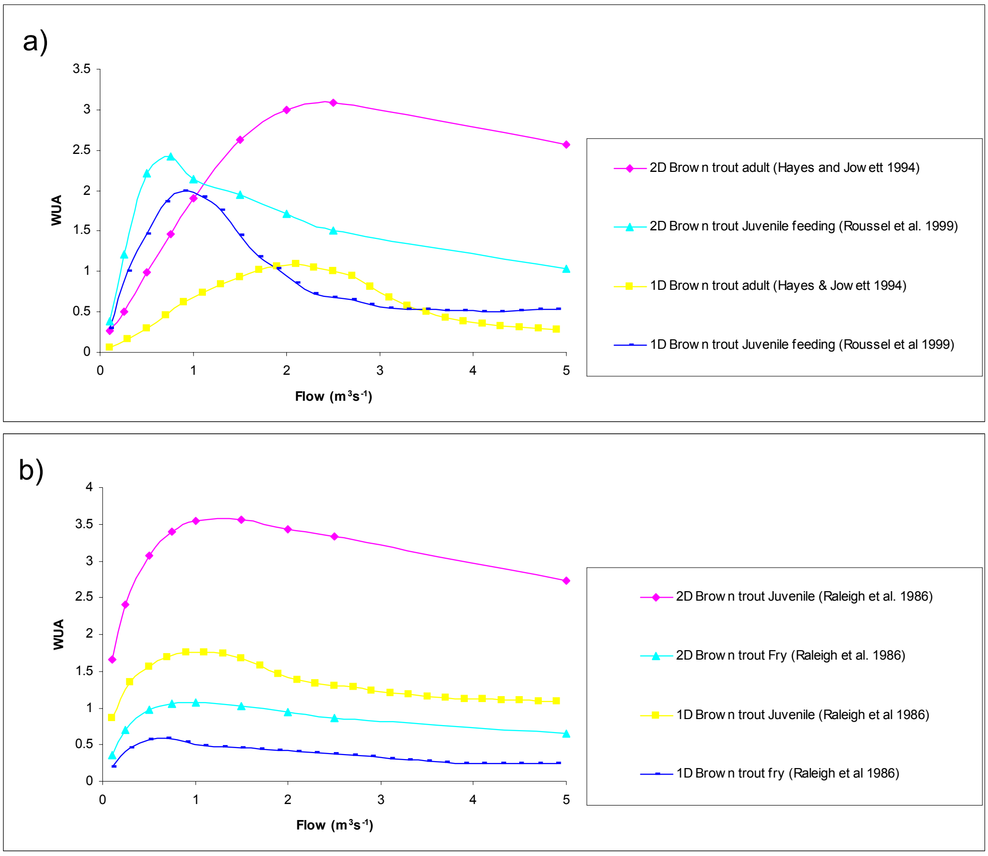


Figure 7. Comparison of WUA (m^2/m) predictions based on 1D and 2D modelling for a range of brown trout life stages in the Motupiko modelled reach over a range of flows. (a) Adult and juvenile feeding habitat, (b) juvenile and fry habitat.

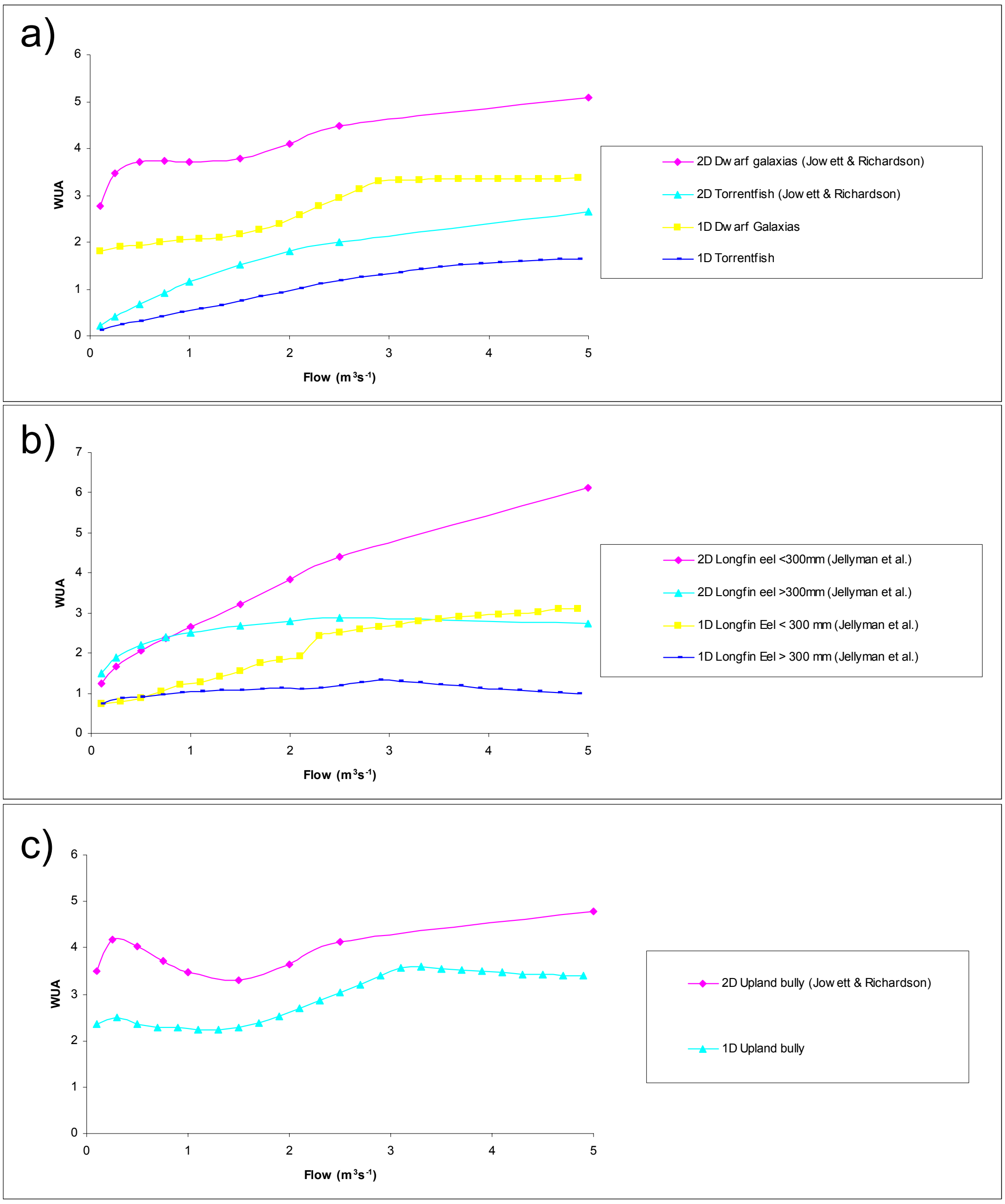


Figure 8. Comparison of WUA predictions based on 1D and 2D modelling for a range of native fish species in the Motupiko modelled reach over a range of flows. (a) Dwarf galaxias and torrentfish, (b) Longfin eel <300 mm and >300 mm, (c) upland bully

As was the case with HSI, the WUA response curve predicted using Roussel et al.'s (1999) suitability criteria for yearling – small adult brown trout peaked at slightly lower flows with the 2D modelling than under the 1D modelling approach (Figure 7a).

4.2.2. Binary suitability and binary threshold

Over the flow range modelled the binary suitability response curve for the majority of species was similar in shape to the expected response of simple hydraulic parameters to increasing flow (Figure 9 c.f. Figure 1). River width, for example, generally increases approximately in proportion to the square root of discharge (with exponents ranging between 0.45-0.54; Jowett 1997). In this case the shape of the binary response curves for most species closely followed the curve describing total wetted area in the reach (i.e. average wetted width x reach length; Figure 9). This suggests that simply classifying any tolerable habitat as suitable essentially reduces this habitat based instream flow method to the equivalent of a more simplistic hydraulic method, at least over this flow range in this reach. Species are often able to tolerate a reasonably broad range of conditions, although their optimum or preferred range of conditions may be quite narrow, so under moderate flow conditions very little wetted habitat is likely be classified as totally intolerable. This appears to render the binary suitability approach more an index of wetted area than an index of the availability of quality habitat, under moderate flow conditions.

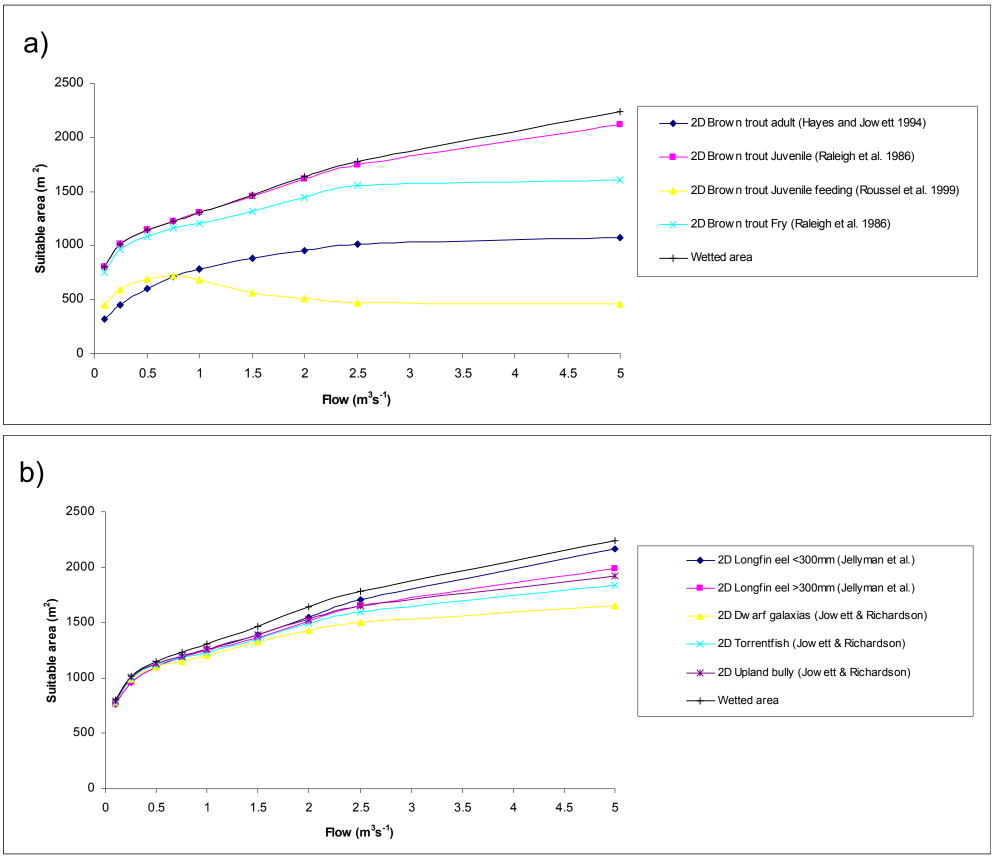


Figure 9. Predictions of binary suitability versus flow based on 2D modelling for a range of brown trout life stages (a) and native fish species (b) in the Motupiko modelled reach.

The main exception in the shape of its binary suitability response curve was yearling – small adult brown trout feeding habitat based on Roussel et al.'s (1999) suitability criteria (Figure 9a). This is probably due to the relatively low and narrow velocity and depth tolerance maxima in these criteria (approximately 0.5 ms^{-1} and 0.5 m respectively, c.f. maximum depth and velocity suitability in other brown trout HSC; Appendix 1), which are probably the result of these curves having been developed on a stream with low discharge. Once the predicted depths and velocities in the modelled reach exceed the maximum tolerable values specified in these criteria the physical habitat is no longer suitable (i.e. the suitability score for any point with excessive depth or velocity is zero). This appears to occur over much of the modelled reach at relatively low flows (approximately 1 m^3s^{-1}) for the Roussel et al. (1999) criteria. Presumably, if high enough flows were modelled, producing sufficiently extreme predicted depths and velocities, then the binary suitability response curves of other species would also begin to show a decline.

Although the binary suitability response curve for adult brown trout based on Hayes & Jowett (1994) followed a similar shape to the wetted area curve (Figure 9a), it reached a much lower magnitude. This is presumably because much of the wetted area was too shallow to be deemed suitable habitat for adult trout based on these HSC (e.g. in the margins). These criteria stipulate a minimum tolerable depth of 0.2 m.

So rather than cancelling out the effect of the suitability criteria altogether, rendering the results entirely dependent on hydraulic geometry (as suggested above), implementing binary suitability criteria in this way actually just reduces the sensitivity of the results to the habitat suitability criteria.

By contrast the binary threshold habitat response is highly sensitive to the suitability criteria. It does arguably provide an indication of the availability of optimal habitat. Comparing the response curves from the standard binary suitability approach with the curves for the binary threshold (Figures 9 and 10), it is apparent that although the area of suitable habitat tends to increase with flow for most species, very little of this habitat is of optimal quality. For those species whose binary threshold response curves do not flat-line, the shape of the response curves were generally similar to the traditional WUA curves, if somewhat more peaky (Figure 10 c.f. Figures 7 and 8).

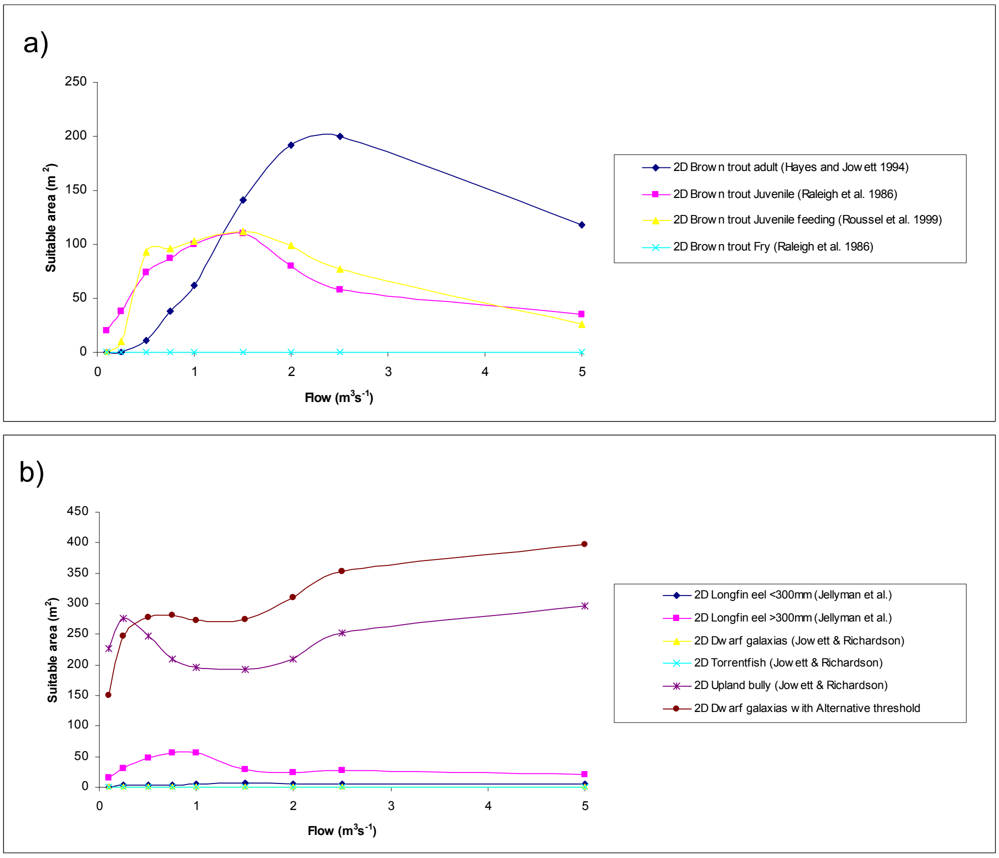


Figure 10. Predictions of binary threshold suitability versus flow based on 2D modelling for a (a) range of brown trout life stages and (b) native fish species in the Motupiko modelled reach.

Again the notable exception to this was yearling – small adult brown trout feeding habitat based on Roussel et al.'s (1999) suitability criteria. The binary threshold response curve based on these criteria peaked at about 1.5 m^3s^{-1} (Figure 10), whereas, all the other metrics based on this set of HSC showed a declining trend as flow increased through this flow (Figures 5a, 7a and 9a). This indicates that although the overall quantity and quality of predicted habitat had begun to decline, the actual area of optimal habitat available continued to increase up to this flow.

One obvious disadvantage of the binary threshold approach is its sensitivity to the arbitrary threshold level selected. This is exemplified by dwarf galaxias in this reach. The dominant substrate type through the majority of the modelled reach was classed as cobbles. The dwarf galaxias suitability criteria had a suitability value of 0.8 for cobbles. So by setting the threshold level to values >0.8 the majority of the reach area was effectively excluded as unsuitable habitat. However, simply changing the threshold level to values ≥0.8 reclassified a large area of previously unsuitable habitat as suitable (Figure 10b 2D Dwarf galaxias with Alternative threshold).

The 2D plan view output plots, showing the spatial distribution of suitable habitat through the modelled reach (e.g. Figure 4b), are often perceived as an advantage of the 2D modelling approach (NB Although it is also possible to produce 2D plan view plots in RHYHABSIM, and presumably in other 1D modelling packages, this is very seldom done). While these plots are informative and can be impressive, it is very difficult to perceive trends in the relationship between habitat and flow when confronted with a large array of these plots representing different flow by species (and lifestage) combinations. Simple line graphs provide a more succinct illustration of these trends (e.g. to illustrate the information on the flow versus habitat relationships for the four life stages of brown trout summarised in Figure 10a, would have required 36 2D plan view plots). So while 2D plan view plots may be useful for some applications (e.g. examining the spatial distribution of suitable habitat through a reach, for comparison with observed fish locations), simple line graphs are arguably better suited to examining habitat versus flow relationships for flow decision making.

5. APPLICATION OF THE 2D METHOD TO THE MOTUEKA AT TAPAWERA

The 2D habitat modelling approach was applied to a reach in the Motueka River upstream of Tapawera (Figure 11), with the aim of providing guidance on a minimum flow requirement to maintain instream habitat for this section of the river.

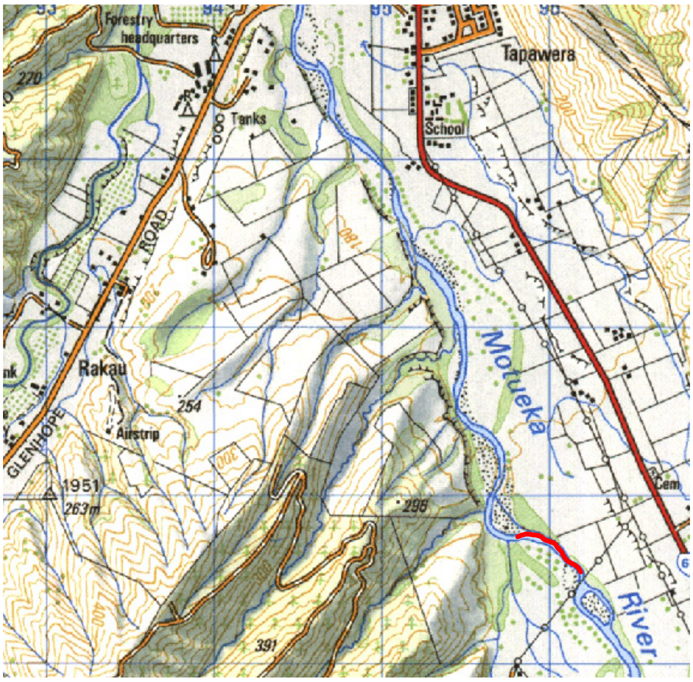


Figure 11. The Motueka River upstream of Tapawera showing the 2D habitat modelling reach (marked in red).

5.1. Methods

5.1.1. Field survey

The reach surveyed on the Motueka River was identified through consultation with Joseph Thomas (TDC). There are substantial losses of surface water to the aquifer in this section of the river and thus this reach was considered to represent the area that will first experience the impacts of low flows. If an appropriate minimum flow is determined for this reach then habitat quality in other parts of this section of the river should also be maintained.

The reach comprised a single riffle, run, pool sequence approximately 400 m long and was surveyed over three days (21, 22 and 27 April 2005).

The field survey involved a similar process to that described for the Motupiko reach (see Section 3.1.1). A topographical survey of the streambed was made using a total station (Trimble 5600-series), resulting in a data set consisting of 953 points to describe the streambed topography of this reach.

As well as the topographical survey, flow was gauged at three cross-sections through the reach. The depth and velocity measurements from these cross-sections were later used to calibrate the flow model. The cross-sections were located to sample the different micro-habitat types that occurred in the reach (i.e. run, riffle, pool).

All flow dependent data (i.e. gaugings, cross-sectional and water level measurements for calibration) were collected on the first two days of the survey to minimise the impact of temporal flow variations on these measurements. Measurements of bed topography were then completed on the subsequent visit.

The flow gauged at the reach at the time of the survey was 2.92 m^3s^{-1}. Additional gaugings and cross-sectional measurements used to validate the model's depth and velocity predictions at a different flow were made on 26 May 2005, at a flow of 3.93 m^3s^{-1}.

5.1.2. The 2D hydraulic modelling process

The 2D flow modelling was undertaken using River2D (Steffler et al. 2003) following the process described in Section 3.1.2.

Topographical survey data were plotted and edited in a triangulated irregular network (TIN).

Bed roughness was specified based on polygons of homogenous substrate size, identified during the field survey. Bed roughness was initially set to two times the largest dimension of the dominant substrate in each of these polygons. This was subsequently manipulated as part of the flow model calibration process, to improve the match between observed and modelled depths, velocities and water surface levels.

A computational mesh composed of 2,964 nodes was developed based on this bed file. This mesh was constructed using the technique recommended by River2D's developers (Waddle & Steffler 2002). Mesh breaklines were defined to coincide with all major topographical breaklines. Additional breaklines were added to divide the flow into at least 8-10 elements across the channel. The inflow and outflow boundaries were subdivided by at least 20 nodes. Nodes were spaced more closely in areas with more complex topography or complex flow features. The contour lines produced based on the bed file and those based on the computational mesh were compared visually to ensure that an adequate description of the bed topography was reproduced. More nodes were added to improve the topographic reproduction as needed.

Two dimensional depth averaged flow through the reach was then modelled using River2D. First, flow solutions for the measured survey flows were calibrated against the observed depth and velocity measurements taken at the cross-sections in the reach, and the modelled wetted perimeter was compared visually with the shape and location of the observed water's edge. Bed roughness values were manipulated (based on the areas identified during the field surveys as having relatively homogeneous substrate size) and the flow model rerun in an attempt to minimise the sum of the squared differences between modelled and observed depths and velocities across the calibration cross-sections. The bed roughness distribution that was developed through this process was then used in subsequent modelling of flows other than the calibration flow.

Unfortunately, water surface elevation (stage height) was not measured at the bottom of the reach during the follow up survey. To facilitate modelling a range of flows it is necessary to have a stage-discharge relationship for the outflow boundary, because a fixed water surface elevation must be specified for each model run as the outflow boundary condition. In order to develop a stage-discharge relationship for this flow boundary we needed a measurement of the stage at the second survey flow (3.93 m^3s^{-1}), in addition to the stage at zero flow estimate and the stage at the first survey flow (2.92 m^3s^{-1}) that we had. To solve this problem we modelled a flow solution for the second survey flow using the same outflow boundary condition as had been used for the first flow. We then queried the model for the predicted water surface elevation at 452 points in a steep riffle immediately upstream of the outflow boundary and compared these predictions with the water surface elevation predicted at these same points in the first survey flow solution. The average difference in water surface elevation between these flow solutions was 14 mm, which was taken to be the difference in stage height at the outflow boundary between the two flows. This provided the third point required to develop a stage-discharge relationship for the outflow boundary, which was used to calculate the outflow water surface elevation at other flows modelled. A range of flows from 0.5 m^3s^{-1} to 15 m^3s^{-1} were modelled to encompass the estimated MALF of 1.50-1.55 m^3s^{-1} (Martin Doyle, TDC, pers. comm.).

5.1.3. *Habitat modelling*

The modelled flow solutions formed the basis for instream habitat analysis, predicting changes in habitat with flow change.

As with the Motupiko, the species for which habitat availability was modelled were identified from fisheries distribution maps, based on the New Zealand Freshwater Fisheries Database (NZFFD), outlined in the Motueka integrated catchment management baseline information report (Basher 2003). Suitability criteria for two different size classes of longfin eels were included. For brown trout three sets of suitability criteria were applied, covering different life stages. The juvenile brown trout feeding suitability criteria (Roussel et al. 1999), applied in the Motupiko, were not applied to this reach because the flow range modelled far exceeded the flow under which these criteria were developed, making their transferability questionable in this case. The other species included were dwarf galaxias, torrentfish and upland bullies.

Based on the discussion in Section 3.2.2 above on the short comings of binary suitability and binary threshold approaches to predicting habitat availability, these were not used as a basis for minimum flow recommendations in this report. However, they are presented for comparison with the WUA and HSI curves in Appendix 2.

5.2. Results

The inflow and predicted outflow balanced to within 1% of the inflow discharge for all modelled flow solutions, except the 10 m^3s^{-1} and 15 m^3s^{-1} solutions where the predicted outflows were low by 3.4% and 9.4% respectively, relative to the inflow. The predicted habitat at these highest flows may therefore be underestimated due to this imbalance.

The predicted habitat optimum for adult brown trout occurred at higher flow than any of the other species or life stages modelled, except longfin eels <300 mm long and torrentfish (Figures 12 and 13). This was true of both WUA (peaking close to 10 m^3s^{-1}) and HSI (which peaked lower at about 4.5 m^3s^{-1}). Despite the differences between the peaks of these two metrics the break points for both were similar, occurring at approximately 4 m^3s^{-1}, with both WUA and HSI declining rapidly below this flow.

The WUA response curves for juvenile brown trout were similar for both yearlings and fry, peaking at approximately 3–3.5 m^3s^{-1} (Figure 12a), and declining quite rapidly as flows fell below 2 m^3s^{-1}. Habitat quality (HSI) optima for these life stages occurred at lower flows than did their WUA optima, at 2 m^3s^{-1} for fry and about 1–1.5 m^3s^{-1} for yearlings (Figure 13a).

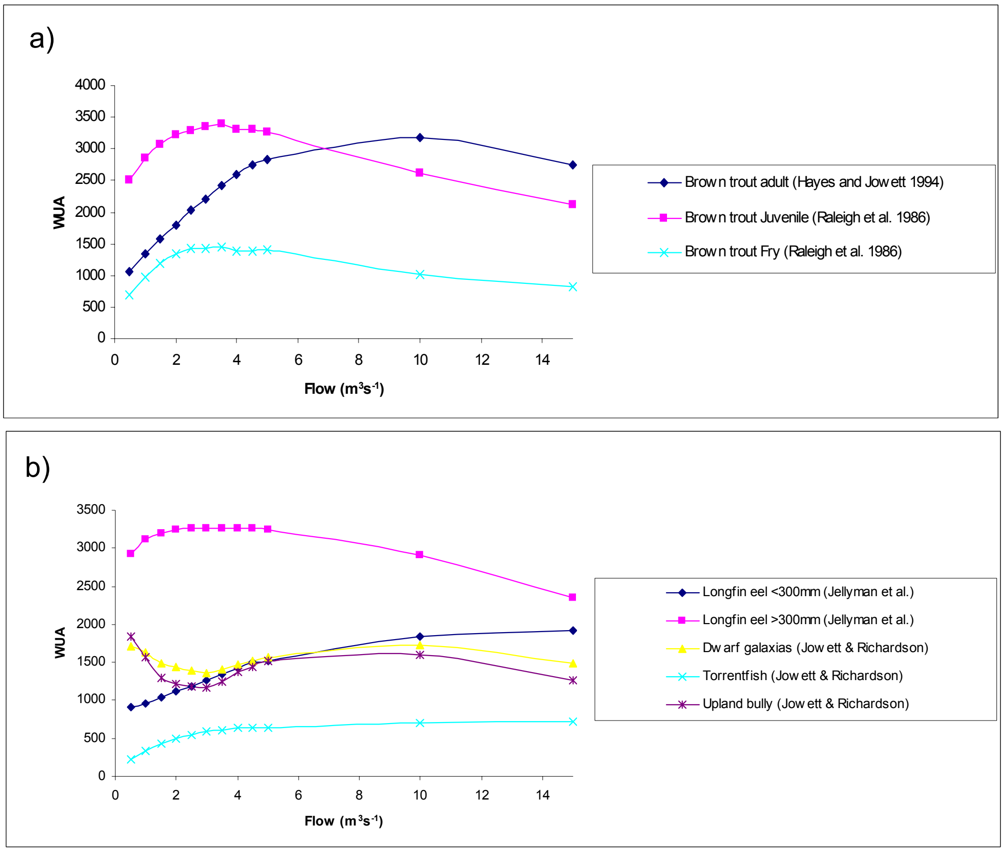


Figure 12. WUA predictions based on 2D modelling for a (a) range of brown trout life stages and (b) native fish species in the Tapawera modelled reach over a range of flows.

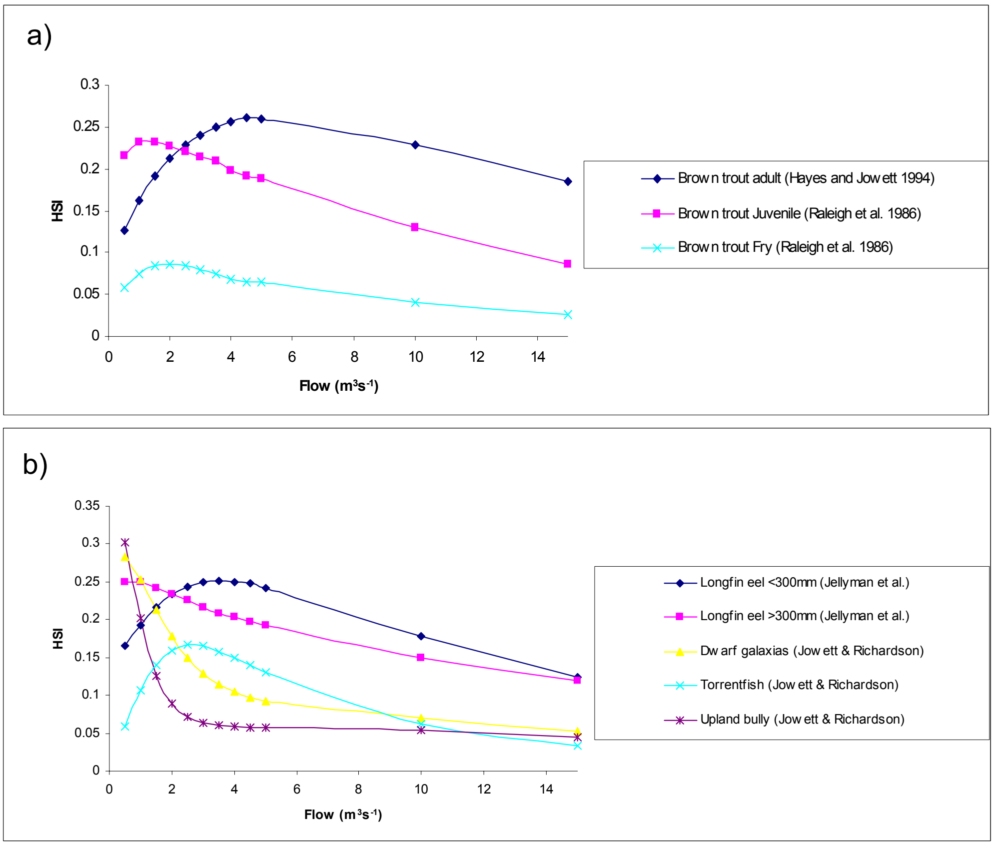


Figure 13. HSI predictions based on 2D modelling for a (a) range of brown trout life stages and (b) native fish species in the Tapawera modelled reach over a range of flows.

WUA was predicted to increase slowly with flow for torrentfish and small longfin eels over the modelled flow range (Figure 12b), but to decrease for larger longfin eels after an initial rise (up until about 3 m^3s^{-1}). However, the average habitat quality (HSI) for torrentfish and small longfin eels was predicted to peak at lower flows, 2.5 m^3s^{-1} and 3.5 m^3s^{-1}, respectively (Figure 13b).

The WUA response for both dwarf galaxias and upland bully showed a similar initial decline with flow but recovered between 3.5 and 5 m^3s^{-1} (Figure 12b), presumably as rising water levels inundated large flat areas of cobble substrate, producing shallow cobble habitat. Habitat quality (HSI) for these two species declined rapidly as flow increased up to approximately 3.5 m^3s^{-1} and more moderately at higher flows (Figure 13b).

5.3. Low flow recommendations for the Tapawera reach

5.3.1. Interpretation of WUA curves for flow management

The insights outlined in Section 2.7 (Ecologically relevant flow statistics for minimum flow setting) have led to a recent move toward interpreting WUA curves in conjunction with flow statistics (notably the MALF) when making decisions on minimum flows (Jowett & Hayes 2004). It has been suggested that if the WUA optima should occur at flows above the MALF, then habitat availability will be limited by the MALF. In this case flow decisions should be made so as to preserve a proportion of the habitat (i.e. WUA) available at the MALF, in order to cater for the needs of both instream values and out-of-stream water uses. In the case where predicted optimum WUA occurs below the MALF, then flows should be managed to maintain a proportion of the habitat available at optimum WUA.

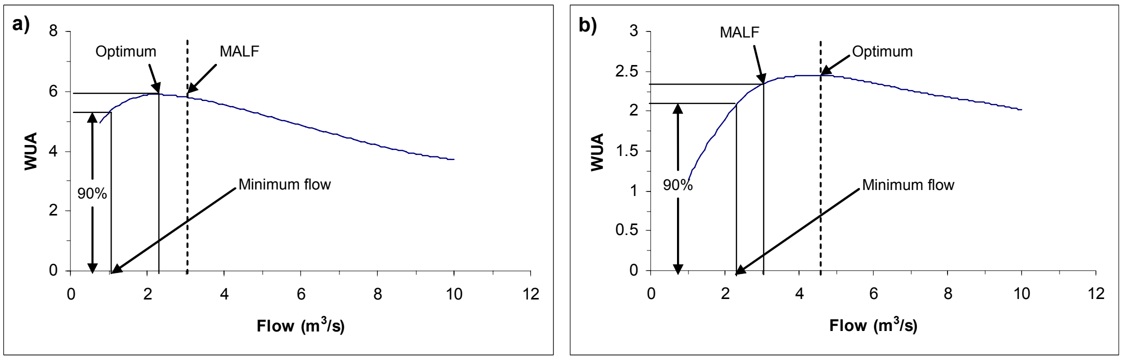


Figure 14. Derivation of minimum flow based on retention of a proportion (90% in this case) of available habitat (WUA) at a) the habitat optimum, or b) the MALF, whichever occurs at the lower flow.

It is then necessary to address how the flow requirements predicted by various WUA versus flow relationships for different species can be reconciled. Jowett & Hayes (2004) suggest that flow dependant critical instream values should be identified and flow decisions made with a focus on managing these values. Candidates for critical value status might include flow sensitive rare or endangered species, or species with high fishery value. "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 also be sustained" (Jowett & Hayes 2004, p. 8). In their document "Flow guidelines for instream values", Ministry for the Environment recommended a similar approach (MfE 1998), although the terminology used differed slightly. Basing decision-making on critical instream values circumvents the complexities of interpreting all the different species' WUA curves independently.

Brown trout provide a highly valued recreational fishery resource in the Motueka River. This suggests trout as a good candidate for critical value status. Trout, especially adult trout, have the highest flow requirements of the species considered in this report (with the possible exception of torrentfish). Therefore, providing for the flow needs of trout will, arguably, provide for the flow needs of less flow-demanding species, as the latter will be able to utilise slower or shallower habitat along the river margins or in riffles.

Of the fish species recorded from the Motueka River in this vicinity both dwarf galaxias and longfin eels are listed as being in "gradual decline" under the New Zealand threat classification system (Hitchmough et al. 2007). As can be seen in the WUA and HSI response curves, habitat for dwarf galaxias and longfin eels >300 mm would be optimised at flows well below what would be required by brown trout, especially adult brown trout. The WUA response of longfin eels <300 mm declines linearly with flow reduction below about 5 m^3s^{-1}, but the rate of decline is not as steep as that for adult brown trout. Consequently, a minimum flow based on retention of a proportion of habitat at the MALF would be more conservative if based on adult brown trout WUA.

The habitat requirements of adult brown trout are arguably the most pertinent to minimum flow setting for this section of the river. It is large adult fish that support the recreational fishery. Also this section has the potential to provide good habitat for large adult trout, with favourable deep pool habitat combined with reasonable food producing riffle habitat. Provision of flows which maintain adult trout habitat should also provide some habitat suitable for other life stages of trout. However, habitat for younger trout is arguably of lesser importance in this reach, since adequate spawning and juvenile rearing habitat is likely to be found in tributaries higher up in the catchment.

Based on the above rationale we recommend that adult brown trout habitat be adopted as the critical instream value, to be maintained during minimum flow decision making, in the section considered here.

Finally, the decision remains as to what level of habitat quantity should be maintained. The level of habitat retention is arbitrary, and scientific knowledge of the response of river ecosystems, and fish populations in particular, is insufficient to identify levels of habitat below which ecological impacts will occur. A carefully designed and well funded monitoring programme might detect effects of a 50% reduction in habitat on fish populations but is unlikely to detect effects of a 10% reduction in habitat – due mainly to the large natural spatial and temporal variability typical of fish populations. It is uncertain whether any effects of a 20% reduction in habitat on fish populations would be detectable.

Jowett & Hayes (2004) recognise that, in practice, the choice of a habitat retention level is based more on risk management than ecological science. The risk of ecological impact increases as habitat is reduced. When instream resource values are factored into the decision making process, then the greater the resource value the less risk is acceptable. With this in mind, Jowett & Hayes (2004) suggest that water managers could consider varying the percent habitat retention level, depending on the value of instream and out-of-stream resources (i.e. highly valued instream resources warrant a higher level of habitat retention than low valued instream resources). This concept is consistent with conservative flow decisions in national water conservation orders (usually no more than 5% habitat reduction). Table 1 shows how Jowett & Hayes (2004) envisage that percentage habitat retention could be varied to take account of variation in instream values.

Table 1. Suggested significance ranking (from highest (1) to lowest (5)) of critical values and levels of habitat retention.

Critical value	**Fishery quality**	**Significance ranking**	**% habitat retention**
Large adult trout – perennial fishery	High	1	90
Diadromous galaxiid	High	1	90
Non-diadromous galaxiid	-	2	80
Trout spawning/juvenile rearing	High	3	70
Large adult trout – perennial fishery	Low	3	70
Diadromous galaxiid	Low	3	70
Trout spawning/juvenile rearing	Low	5	60
Redfin/common bully	-	5	60

Table taken from Jowett & Hayes (2004)

Applying these criteria to the Motueka River, with its highly valued fishery resource, would suggest a minimum flow that retains 90% of the predicted habitat available for adult brown trout at the MALF. This would give a minimum flow of 1.2 m^3s^{-1} for this reach (Table 2).

Table 2. Suggested minimum flow options for the reach of the Motueka River upstream of Tapawera, based on retention of a percentage of the habitat availability (WUA) predicted at the MALF with a 2D habitat modelling approach

MALF (m^3s^{-1})	Suitability Criteria	Flow at WUA Optimum (m^3s^{-1})	Flow at 90% of MALF or WUA Optimum (Which ever is the lesser) (m^3s^{-1})	Flow at 80% of MALF or WUA Optimum (Which ever is the lesser) (m^3s^{-1})
1.55	Brown trout adult (Hayes & Jowett 1994)	≈ 10	**1.2**	**0.9**
	Brown trout yearling 15 - 23 cm (Raleigh et al 1986)	3.5	0.9	< 0.5
	Brown trout fry < 14 cm (Raleigh et al 1986)	3.5	1.3	1.0
	Longfin eel <300mm (Jellyman et al. 2003)	> 15	0.7	< 0.5
	Torrentfish (Jowett & Richardson 1995)	> 15	1.3	1.1

However, the section of the Motueka River upstream of Tapawera, where the modelled reach was located, does not support as highly valued a fishery as reaches downstream (e.g. around Woodstock). Consequently, a lower level of habitat retention could be argued for at this reach.

Retention of 80% of the predicted habitat available at the MALF in this reach would require a minimum flow of 0.9 m^3s^{-1} (Table 2).

Whichever level of habitat retention is applied, consideration will have to be given to how these minimum flows will fit with the conditions of the Motueka River Water Conservation Order (2004). Clause 9c of this order prohibits consents or rules in regional plans that "will cause, either by [themselves] or in combination with any other existing consents or rules, alteration of the flow of that part of the Motueka River specified [between the Shaggery River and Wangapeka River confluences]… by more than 12% as measured by the residual flow at Woodstock."

Given that these predictions of the likely response of habitat to changes in flow were based on a single run, riffle, pool sequence, it may be prudent to err on the side of caution when considering minimum flow options for this reach. The 90% habitat retention level would provide a more conservative level of protection for instream values, than the lower level of habitat retention. Also, the suggested minimum flows provided here were based on retention of WUA, rather than HSI, since the former index usually peaks at higher flows and flow decisions based on WUA curves are therefore likely to be more conservative. However, in this case the minimum flows based on adult brown trout habitat retention would have been the same whether they were based on retention of WUA or HSI.

As discussed above it is generally recognised that minimum flows must be set in conjunction with appropriate allocation rules to ensure that a degree of the natural flow variability is maintained, to help maintain invertebrate productivity and other ecological processes. One approach to setting allocation limits, in conjunction with minimum flows, involves defining a "management flow", based on consideration of historic flow frequency and duration data. The historic frequency of occurrence of the "management flow" indicates the expected frequency of occurrence of the minimum flow under the influence of allocation assuming the allocated flow is fully abstracted. Put another way the management flow (and therefore the core allocation) can be set taking into account the acceptable level of risk to the environment and to resource users of the minimum flow occurring. The amount of water available for allocation is then derived from:

Core Allocation = Management Flow – Minimum Flow

This approach has been employed by Horizons Regional Council in setting allocation limits in rivers in their jurisdiction.

A potential alternative to this method is outlined in Jowett & Hayes (2004, Section 6 Total allocation). This method is based on balancing the level of total allocation with the minimum flow applied, such that abstraction should not increase the amount of time that the river is at or below the minimum flow by more than some predetermined percentage.

We suggest that another factor that could be considered in this process is to ensure that the invertebrate habitat at the median flow is not reduced excessively by water allocation. This would provide a biological rationale for the level of allocation in addition to that underpinning the setting of the minimum flow. Invertebrate habitat at the median flow is relevant to maintenance of the productivity of invertebrate populations, which provide the food base for fish. If they are not limited by habitat availability, then fish populations are likely to be limited either by flood disturbance, or by food availability; it is a central tenet of ecology that populations must ultimately be limited by some factor, or else they would continue to grow to become infinitely large. Large scale disturbance caused by floods is unlikely to be altered significantly by flow management except in extreme cases where large dams are capable of capturing large flood flows. However, moderate to large scale water abstraction can alter flow regimes sufficiently to potentially impact on food availability by temporarily reducing invertebrate habitat with associated reduction in invertebrate production. Generally optimal invertebrate habitat occurs at higher flows than optimal fish habitat and because they have high rates of colonisation invertebrates can make productive use of extended flow recessions. For instance, they take about 30 days to fully colonise previously dry channels (or margins) (Sagar 1983).

6. WHERE TO FROM HERE?

This report demonstrates many of the options available for relating flow to instream habitat quality and provides guidance on an appropriate minimum flow for the Motueka River upstream of Tapawera.

All of these modelling efforts are based at a reach scale and therefore the information is really only applicable to sections of the Motueka River which are represented by the reaches where field surveys and modelling have been conducted. Ideally, it would be useful to look at flow management in the catchment as a whole. Over the next few years we will be attempting to scale-up some of this reach scale information to better understand how habitat availability and the carrying capacity for fish changes with flow throughout the catchment. This will be achieved by conducting field surveys on key sections of the river that are not represented by existing surveys. We will also be incorporating principles from energetics-based models (Hayes et al. 2000, 2003) to allow a mechanistic understanding of the effects of flow, channel morphology, water temperature, and food availability on habitat quality throughout the catchment.

7. REFERENCES

Addley RC 1993. A mechanistic approach to modeling habitat needs of drift feeding salmonids. MS thesis. Utah State University, Logan, Utah.

Addley RC 2006. Habitat modelling of river ecosystems: multidimensional spatially explicit and dynamic habitat templates at scales relevant to fish. PhD thesis, Utah State Univeristy, Logan, Utah.

Allan JD 1995. Stream Ecology: Structure and function of running waters. Kluwer Academic Publishers. Dordrecht, The Netherlands.

Annear T, and 15 other authors 2002. Instream flows for riverine resource stewardship. Instream Flow Council, US.

Bartholow JM 1989. Stream temperature investigations: field and analytic methods. Instream flow information paper 13. U.S. Fish and Wildlife Service Biological Report 89(17). 139 p.

Basher LR comp. 2003. The Motueka and Riwaka Catchments: A technical report summarising the present state of knowledge of the catchment, management issues and research needs for integrated catchment management. Landcare Research New Zealand, Lincoln, Cantebury, NZ. 120 p.

Beffa C 1996. Application of a shallow water model to braided flows. In: Mueller A ed. Proceedings of the Hydroinformatics 96 Conference. Pp. 667-672.

Bell VA, Elliott JM, Moore RJ 2000. Modelling the Effects of Drought on the Population of Brown Trout in Black Brows Beck. Ecol.Model. 127: [2-3], Pp. 141-159.

Bovee KD 1982. A guide to stream habitat analysis using the instream flow incremental methodology. U.S. Fish and Wildlife Service Biological Services Program FWS/OBS-82/26, Instream flow information paper 12. 248 p.

Castleberry DT, Cech JJ, Erman DC, Hankin D, and others 1996. Uncertainty and instream flow standards. Fisheries 21(8): 20-21.

Chapman DW 1966. Food and space as regulators of salmonid populations in streams. The American Naturalist 100: 346-357.

Collier KJ, Wakelin MD 1995. Instream habitat use by blue duck on Tongariro River. National Institute of Water and Atmospheric Research, Hamilton, NIWA Science and Technology 28. 35 p.

Denslinger TL, Gast WA, Hauenstein JJ, Heicher DW, Henriksen J, Jackson DR, Lazorchick GJ, McSparran JE, Stoe TW, Young LM 1998. Instream flow studies Pennsylvania and Maryland. Susquehanna River Basin Commission Publication No. 191. Prepared in cooperation with the Pennsylvania Department of Environmental Protection under Contract ME94002.

Dunbar MJ, Gustard A, Acreman MC, Elliot CRN 1998. Overseas approaches to setting River Flow Objectives. R&D Technical Report W6-161. Environment Agency and Institute of Hydrology, Wallingford. 83 p.

Duncan MJ 1992. Flow regimes of New Zealand rivers. Pp. 13-27 In: Waters of New Zealand. MP Mosley Ed. New Zealand Hydrological Society, Wellington.

Duncan MJ, Carter GC 1997. Two-dimensional hydraulic modelling of New Zealand Rivers: the NIWA experience. 24th Hydrology & Water Resources Symposium Proceedings: 493-497.

Duncan MJ, Hicks DM 2001. 2-D habitat modelling for the Rangitata River. Unpublished NIWA Client Report CHC01/72, NIWA, Christchurch.

Elliott JM 1994. Quantitative ecology and the brown trout. Oxford. Oxford University Press.

Elliott JM, Hurley MA, Elliott JA 1997. Variable Effects of Droughts on the Density of a Sea-Trout Salmo Trutta Population Over 30 Years. Journal of Applied Ecology 34: 1229-1238.

Fraser JC 1978. Suggestions for developing flow recommendations for instream uses of New Zealand streams. Water and Soil Miscellaneous Publication 6, Ministry of Works and Development, Wellington.

Gallagher SP, Gard MF 1999. Relation between chinook salmon (*Oncorhynchus tshawtscha*) redd densities and PHABSIM predicted habitat in the Merced and Lower American Rivers, CA. Canadian Journal of Fisheries and Aquatic Sciences 56: 570-577.

Ginot V 1998. Logiciel EVHA. Evaluation de l'habitat physique des poissons en rivière (version 2.0). Cemagref Lyon BEA/LHQ et Ministère de l'aménagement du Territoire et de l'Environnement, Direction de l'Eau, Paris, France.

Gore JA, Nestler JM 1988. Instream flow studies in perspective. Regulated Rivers: Research and Management 2: 93-101.

Guay JC, Boisclair D, Rioux D, Leclerc M, Lapointe M, Legendre P 2000. Development and validation of numerical habitat models for juveniles of Atlantic salmon (*Salmo salar*). Canadian Journal of Fisheries and Aquatic Sciences 57: 2065-2075.

Guay JC, Boisclair D, Leclerc M, Legendre P 2001. Science on the edge of spatial scales: a reply to the comments of Williams (2001). Canadian Journal of Fisheries and Aquatic Sciences 58: 2108-2111.

Guensch GR, Hardy TB, Addley RC 2001. Examining feeding strategies and position choice of drift-feeding salmonids using an individual-based, mechanistic foraging model. Canadian Journal of Fisheries and Aquatic Sciences 58: 446-457.

Hatfield T, Bruce J 2000. Predicting salmonid habitat–flow relationships for streams from western North America. North American Journal of Fisheries Management 20: 1005-1015.

Hayes JW 1998. Riwaka River minimum flow study. Prepared for Tasman District Council. Cawthron Report No. 400. 19 p. plus appendices.

Hayes JW 2002. Evidence of John William Hayes before the Environment Court in the matter of an inquiry into the draft National Water Conservation (Motueka River) Order 1991. 19 p.

Hayes JW 2003. Maitai River instream habitat flow analysis. Prepared for Nelson City Council. Cawthron Report No. 795. 22 p.

Hayes JW, Jowett IG 1994. Microhabitat models of large drift-feeding brown trout in three New Zealand rivers. North American Journal of Fisheries Management 14: 710-725.

Hayes JW, Hughes NF, Kelly LH 2003. Overview of a process-based model relating stream discharge to the quantity and quality of brown trout habitat. Proceedings of the International IFIM Workshop. Fort Collins, Colorado, June 2003.

Hayes JW, Stark JD, Shearer KA 2000. Development and test of a whole-lifetime foraging and bioenergetics growth model for drift-feeding brown trout. Transactions of the American Fisheries Society 129: 315-332.

Heggenes J 1988. Physical habitat selection by brown trout (*Salmo trutta*) in riverine systems. Nordic Journal of Freshwater Research 64: 74-90.

Heggenes J 1996. Habitat selection by brown trout (*Salmo trutta*) and young atlantic salmon (*S. salar*) in streams: static and dynamic hydraulic modelling. Regulated Rivers: Research & Management 12: 155-169.

Hill J, Grossman GD 1993. An energetic model of microhabitat use for rainbow trout and royside dace. Ecology 7: 685-698.

Hitchmough R, Bull L, Cromarty P comps 2007. New Zealand Threat Classification System lists—2005. Department of Conservation, Wellington. 194 p.

Hudson HH, Byrom AE, Chadderton LW 2003. A critique of the IFIM – instream habitat simulation in the New Zealand context. Department of Conservation, Science for Conservation 231.

Hudson HR, McMillan DA, Grey A 2005. Proposed Wairau Valley Hydroelectric Power Scheme Scope 1: Hydraulic and habitat simulation, Wairau River, Marlborough. EMA 2005-05, Environmental Management Associates, Christchurch, 93 p.

Hughes NF, Dill LM 1990. Position choice by drift-feeding salmonids: Model and test for Arctic grayling (*Thymallus arcticus*) in subarctic mountain streams, interior Alaska. Canadian Journal of Fisheries and Aquatic Sciences 47: 2039-2048.

Hughes NF, Hayes JW, Shearer KA, Young RG 2003. Testing a model of drift-feeding using 3-D videography of wild brown trout in a New Zealand river. Canadian Journal of Fisheries and Aquatic Sciences 60: 1462-1476.

Jellyman DJ, Bonnett ML, Sykes JRE, Johnstone P 2003. Contrasting use of daytime habitat by two species of freshwater eel (Anguilla spp) in New Zealand rivers. In: Dixon DA ed. Biology, Management and Protection of Catadromous Eels. American Fisheries Society Symposium 33, Bethesda, Maryland, USA.

Jorde K 1997. Ökologisch begründete, dynamische Mindestwasserregelungen bei Ausleitungskraftwerken. Mitteilungen des Instituts für Wasserbau, Heft 90, Universität Stuttgart, 155 p.

Jowett IG 1989. River hydraulic and habitat simulation, RHYHABSIM computer manual. New Zealand Fisheries Miscellaneous Report 49. Ministry of Agriculture and Fisheries, Christchurch. 39 p.

Jowett IG 1990. Factors related to the distribution and abundance of brown and rainbow trout in New Zealand clear-water rivers. New Zealand Journal of Marine and Freshwater Research 24:429-440.

Jowett IG 1992. Models of the abundance of large brown trout in New Zealand rivers. North American Journal of Fisheries Management 12: 417-432.

Jowett IG 1993a. Minimum flow assessments for instream habitat in Wellington rivers. NZ Freshwater Miscellaneous Report No. 63. National Institute of Water and Atmospheric Research, Christchurch. 33 p.

Jowett IG 1993b. Report on minimum flow requirements for instream habitat in Taranaki rivers. NZ Freshwater Miscellaneous Report No. 75. National Institute of Water and Atmospheric Research, Christchurch. 35 p.

Jowett IG 1994. Minimum flows for native fish in the Waipara River. NIWA Christchurch Miscellaneous Report No. 180. 21 p.

Jowett IG 1997. Instream flow methods: a comparison of approaches. Regulated Rivers 13: 115-127.

Jowett IG 1998. Hydraulic geometry of New Zealand rivers and its use as a preliminary method of habitat assessment. Regulated Rivers 14: 451-466.

Jowett IG 1999. WAIORA - interactive technical support for water allocation decisions. Proceedings of the 3rd International Ecohydraulics Conference, Salt Lake City, Utah, IAHR.

Jowett IG 2000. Flow management. In: Collier KJ, Winterbourn MJ eds. New Zealand stream invertebrates: ecology and implications for management. Hamilton, New Zealand Limnological Society. Pp. 289-312

Jowett IG 2002. Instream habitat suitability criteria for feeding inanga (*Galaxias maculatus*). New Zealand Journal of Marine and Freshwater Research 36: 399-407.

Jowett IG 2004. RHYHABSIM River Hydraulics and Habitat Simulation (Software Manual).

Jowett IG, Hayes JW 2004. Review of methods for setting water quantity conditions in the Environment Southland draft Regional Water Plan. NIWA Client Report: HAM2004-018. Prepared for Environment Southland, under NIWA Project: ENS04202.

Jowett IG, Richardson J 1995. Habitat preferences of common, riverine New Zealand native fishes and implications for flow management. New Zealand Journal of Marine and Freshwater Research 29: 13-23.

Kelly L, Hay J, Hughes N, Hayes J 2005. Flow related model for simulating river hydraulics, invertebrate drift transport, and foraging energetics of drift-feeding salmonids: Users guide (Version 1.0). Cawthron Report No. 922. 40 p.

Killingtviet Å, Harby A 1994. Multi-purpose planning with the river system simulator – a decision support system for water resources planning and operation. In: Proceedings of the first international symposium on habitat hydraulics, Norwegian Institute of Technology, Trondheim.

Kondolf GG, Larsen EW, Williams JG 2000. Measuring and modelling the hydraulic environment for assessing instream flows. North American Journal of Fisheries Management 20: 1016-1028.

Lamouroux N, Capra H 2002. Simple predictions of instream habitat model outputs for target fish populations. Freshwater Biology 47: 1543-1556.

Lamouroux N, Capra H, Pouilly M 1998. Predicting habitat suitability for lotic fish: linking statistical hydraulic models with multivariate habitat use models. Regulated Rivers: Research and Management 14: 1-11.

Lamouroux N, Jowett IG 2005. Generalized instream habitat models. Canadian Journal of Fisheries and Aquatic Sciences 62: 7-14.

Leopold LB, Wolman MG, Miller JP 1964. Fluvial processes in geomorphology. San Francisco, W.H. Freeman. 522 p.

Mathur D, Bason WH, Purdy EJ Jr, Silver CA 1985. A critique of the instream flow incremental methodology. Canadian Journal of Fisheries and Aquatic Sciences 42: 825-831.

Milhous RT, Updike MA, Schneider DM 1989. Physical Habitat Simulation System Reference Manual - Version II, U.S. Fish and Wildlife Service Instream Flow Information Paper No. 26. Biological Report 89(16).

Ministry for the Environment 1998. Flow guidelines for instream values. Ministry for the Environment, Wellington.

Morhardt JE, Mesick CF 1988. Behavioral carrying capacity as a possible short-term response variable. Hydro Review 7(2): 32-40.

Mosley MP 1983. Flow requirements for recreation and wildlife in New Zealand rivers – a review. Journal of Hydrology (NZ) 22: 152-174.

Mosley MP, Jowett IG 1985. Fish habitat analysis using river flow simulation. New Zealand Journal of Marine and Freshwater Research 19: 293-309.

Payne TR, Eggers SD, Parkinson DB 2004. The number of transects required to compute a robust PHABSIM habitat index. Hydroécologie appliquée 14: 27-53.

Railsback SF, Dixon DA 2003. Individual-based fish models: ready for business in hydro licensing?. Hydro Review XXII(4): 90-97.

Raleigh RF, Zuckerman LD, Nelson PC 1986. Habitat suitability index models and instream flow suitability curves: brown trout, revised. U.S. Fish and Wildlife Service Biological Report 82 (10.124).

Reiser DW, Wesche TA, Estes C 1989. Status of instream flow legislation and practices in North America. Fisheries 14: 22-29.

Richardson J, Jowett IG 2006. Testing habitat-based in-stream flow methods for native fish. NIWA Poster presented at the New Zealand Freshwater Sciences Society Conference 27-30 November 2006, Rotorua.

Roussel JM, Bardonnet A, Claude A 1999. Microhabitats of brown trout when feeding on drift and when resting in a lowland salmonid brook: effects on Weighted Usable Area. Arch Hydrobiol 146: 413-429.

Sagar PM 1983. Invertebrate recolonisation of previously dry channels in the Rakaia River. New Zealand Journal of Marine and Freshwater Research 17: 377-386.

Scott D, Shirvell CS 1987. A critique of the instream flow incremental methodology and observations on flow determination in New Zealand. In: Kemper JB, Craig J eds. Regulated streams – advances in ecology. Plenum Press, New York. Pp. 27-44.

Singh KP, Broeren SM 1989. Hydraulic geometry of streams and stream habitat assessment. Journal of Water Resources Planning and Management 115(5): 583-597.

Stalnaker CB, Lamb L, Henriksen J, Bovee K, Bartholow J 1995. The instream flow incremental methodology: a primer for IFIM. National Biological Service, Fort Collins, Biological Report 29. 45 p.

Steffler P, Blackburn J 2002. River2D, Two-dimensional depth averaged model of river hydrodynamics and fish habitat: Introduction to depth averaged modeling and user's manual. University of Alberta, Canada. 119 p.

Steffler P, Ghanem A, Blackburn J 2003. River2D Version 0.90 – computer program. University of Alberta, Canada. http://www.River2D.ualberta.ca/index.htm

Tarbet KL, Hardy TB 1996. Evaluation of One-Dimensional and Two-Dimensional Hydraulic Modeling in a Natural River and Implications. In: Leclerc M, Capra H, Valentin S, Boudreault A, Cote Y eds. Proceedings of Ecohydraulics 2000, the Second International Symposium on Habitat Hydraulics, Vol. B, pp. B395-B406. INRS-Eau, FQSA & IAHR/AIRH, Quebec, Canada.

Tennant DL 1976. Instream flow regimens for fish, wildlife, recreation, and related environmental resources. In: Orsborn JF, Allman CH eds, Proceedings of the symposium and speciality conference on instream flow needs II. American Fisheries Society, Bethesda, Maryland. Pp. 359-373.

Tharme RE 1996. Review of international methodologies for the quantification of the instream flow requirements of rivers. Water Law Review Report for Policy Development, Freshwater Research Unit, Zoology Department, University of Cape Town, South Africa.

Thorpe JE 1994. Salmonid Flexibility - Responses to Enviromental Extremes. Transaction of the American Fisheries Society 123: 606-612.

Waddle T, Steffler P 2002. R2D_Mesh, Mesh generation program for River2D two dimensional depth averaged finite element: Introduction to mesh generation and user's manual. U.S. Geological Survey. 31 p.

Waters BF 1976. A methodology for evaluating the effects of different streamflows on salmonid habitat. In: Orsborn JF, Allman CH eds. Proceedings of the symposium and speciality conference on instream flow needs II. American Fisheries Society, Bethesda, Maryland. Pp 224-234.

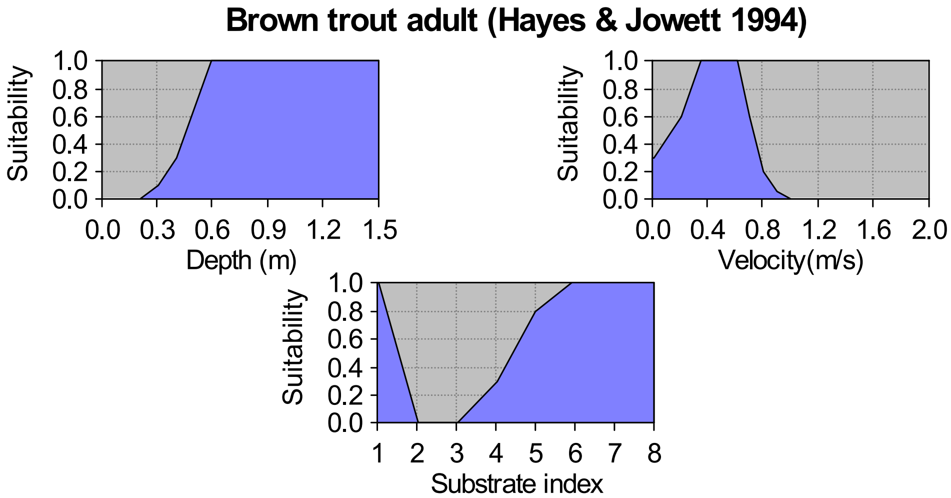
Williams JG 2001. Tripping over spatial scales: a comment on Guay et al. (2000). Canadian Journal of Fisheries and Aquatic Sciences 58: 2105-2107.

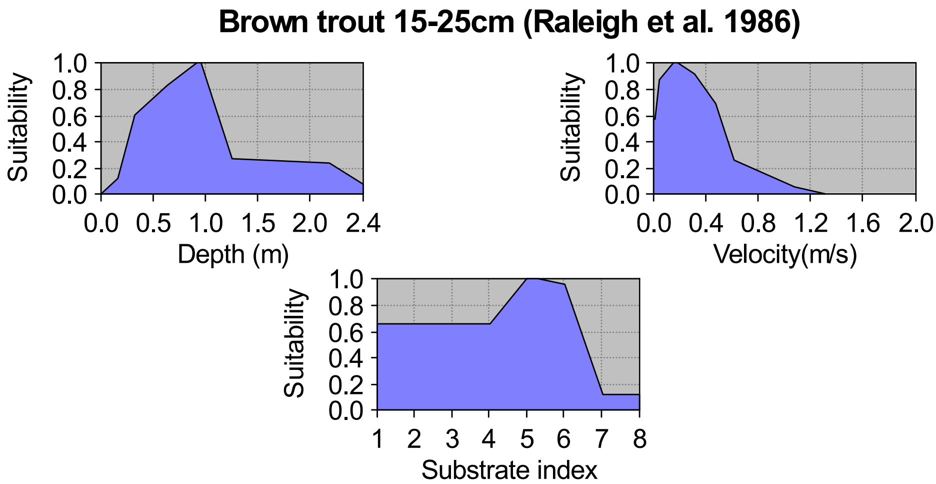
Young RG, Hayes JW 2002. Assessment of some potential techniques to guide management of water abstraction from small streams. Cawthron Report No. 743. Prepared for Tasman District Council, Fish & Game New Zealand, and the Motueka Integrated Catchment Management Program. 18 p.

8. APPENDICES

Appendix 1. Habitat suitability criteria used in this report.

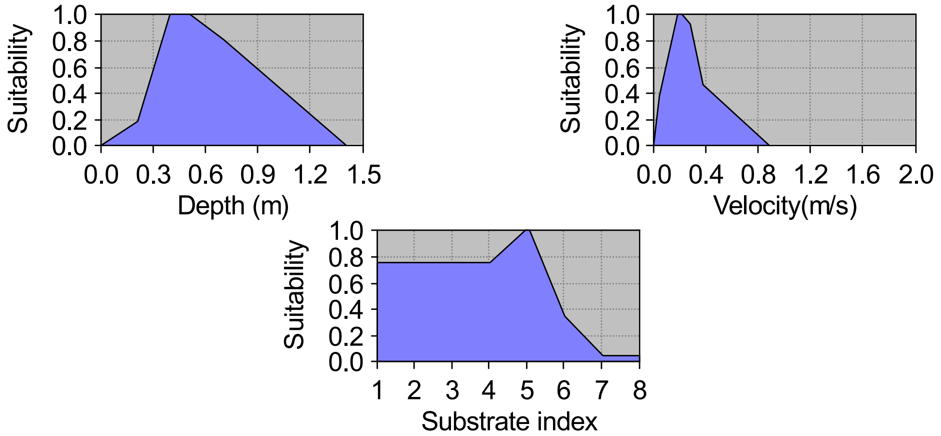
NB Where no citation is given for native fish suitability criteria they are assumed to be based on Jowett & Richardson (1995).



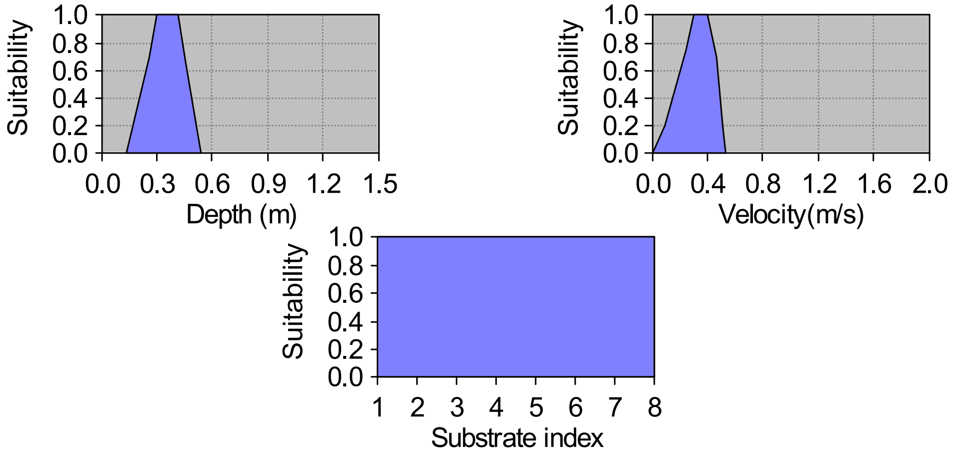


CAWTHRON

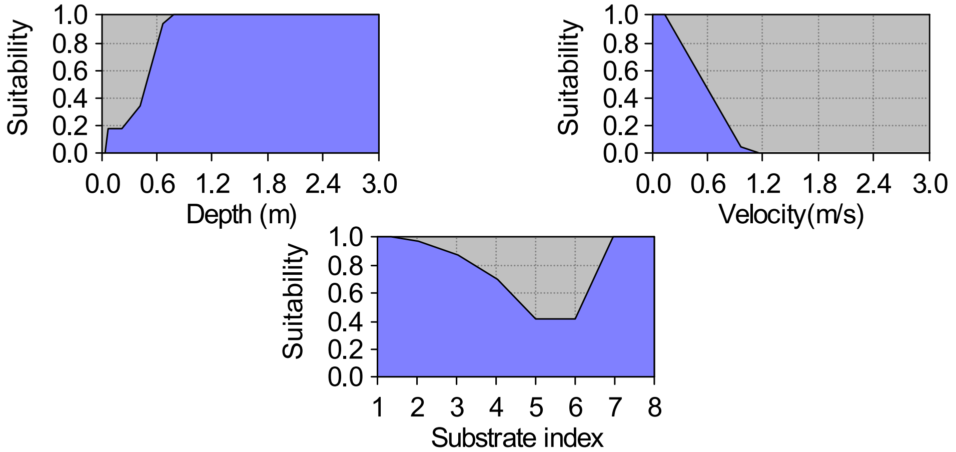
Brown trout fry to 15cm (Raleigh et al. 1986)

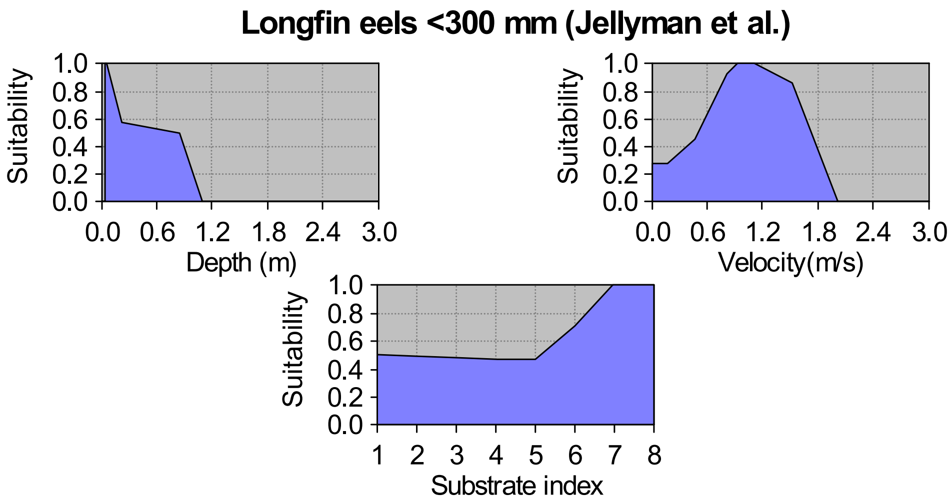


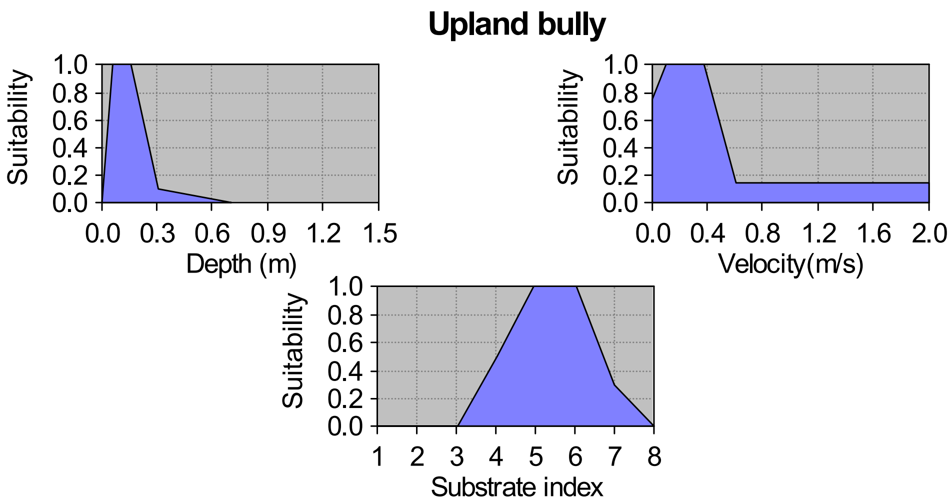
Brown trout yearling - small adult feeding (Roussel et al 1999)



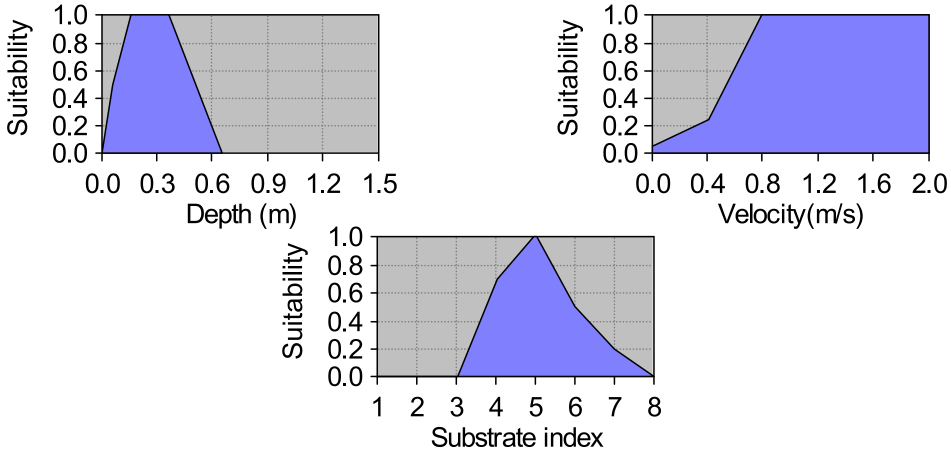
Longfin eels >300 mm (Jellyman et al.)



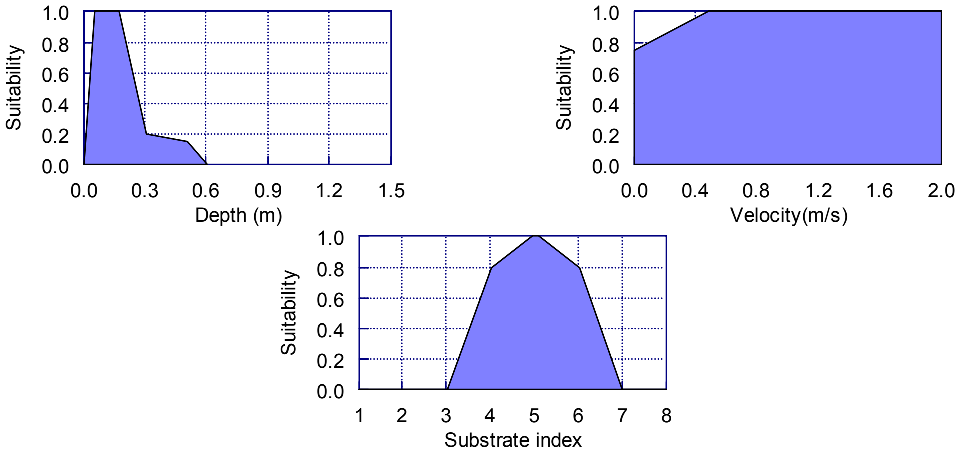

Longfin eels <300 mm (Jellyman et al.)
Suitability
1.0
0.8
0.6
0.4
0.2
0.0
0.0 0.6 1.2 1.8 2.4 3.0
Depth (m)
Velocity(m/s)
1 2 3 4 5 6 7 8
Substrate index


Upland bully
Suitability
1.0
0.8
0.6
0.4
0.2
0.0
0.0 0.3 0.6 0.9 1.2 1.5
Depth (m)
0.0 0.4 0.8 1.2 1.6 2.0
Velocity(m/s)
1 2 3 4 5 6 7 8
Substrate index

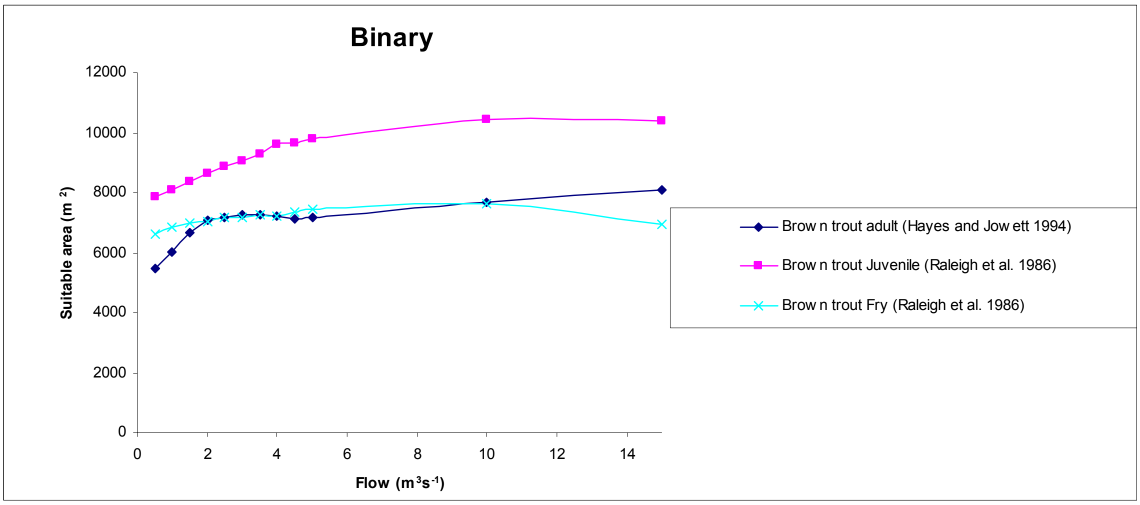
Torrentfish

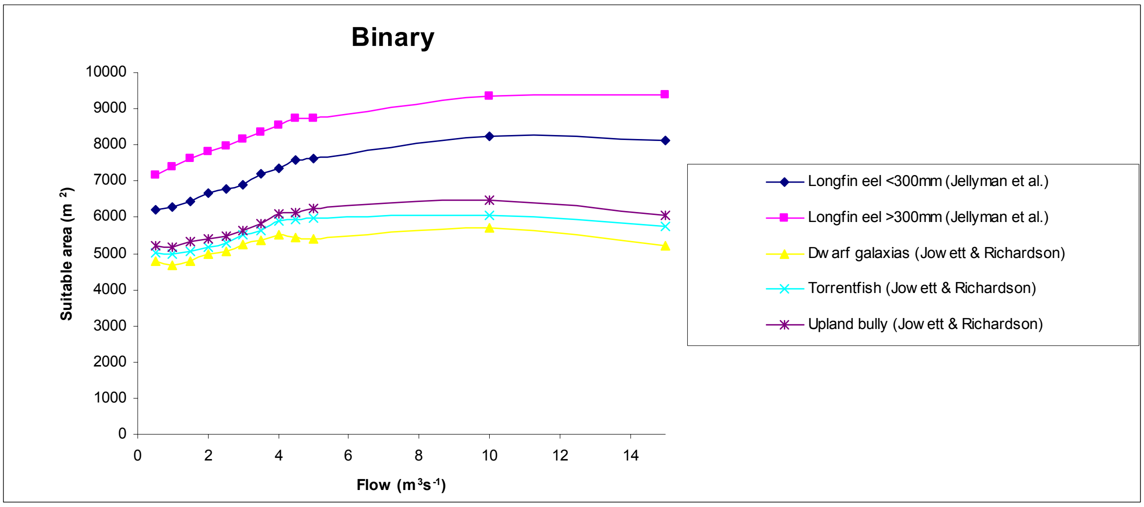


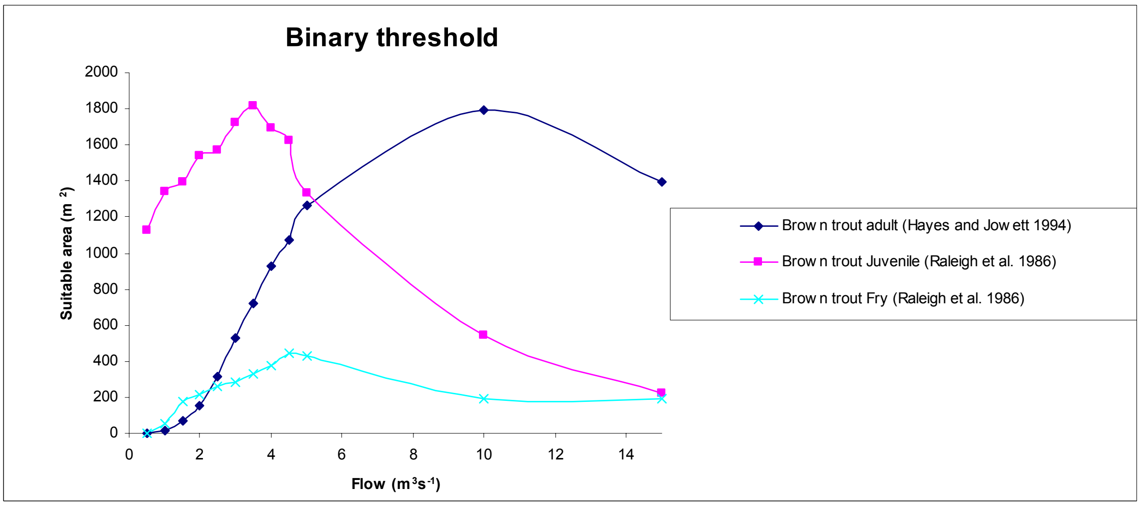
Dwarf Galaxias

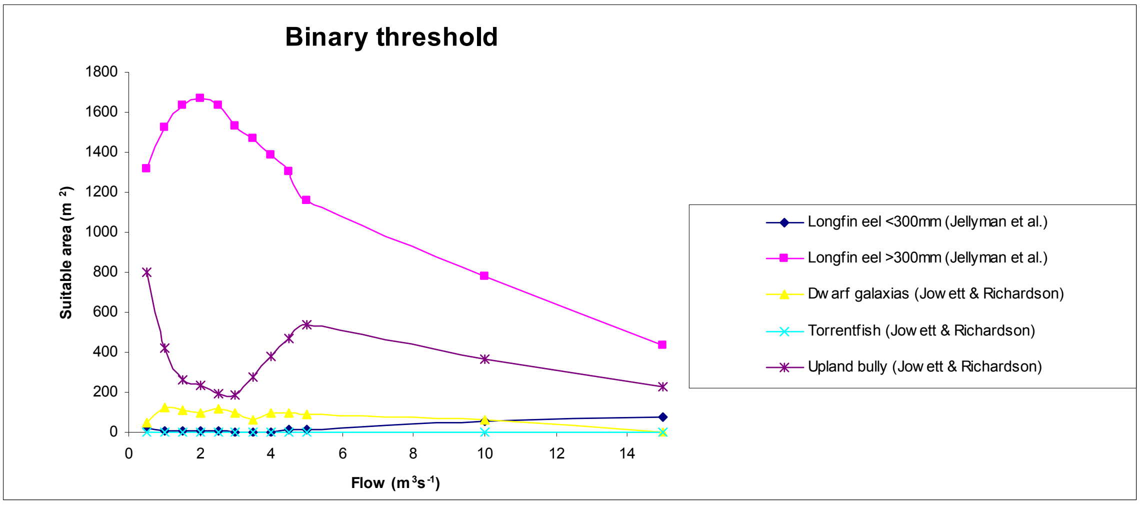


Appendix 2. Predicted habitat responses in the Tapawera reach using the Binary and Binary threshold approaches








Binary threshold
Suitable area (m 2)
1800
1600
1400
1200
1000
800
600
400
200
0
0
2
4
6
8
10
12
14
Flow (m3s-1)
Longfin eel <300mm (Jellyman et al.)
Longfin eel >300mm (Jellyman et al.)
Dwarf galaxias (Jowett & Richardson)
Torrentfish (Jowett & Richardson)
Upland bully (Jowett & Richardson)