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Instream Flow Assessment Options for Horizons Regional Council

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Instream Flow Assessment Options for Horizons Regional Council

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EXECUTIVE SUMMARY

This report addresses options available to Horizons Regional Council for setting defensible minimum flows for rivers in its jurisdiction. It was commissioned to:

1. Summarise methods available for setting minimum flows.
2. Suggest which of these methods (or which combination of methods) could be applied to minimum flow setting within the Horizons region; including verification of the applicability of recently developed generalised habitat models to the Manawatu/Wanganui region (by comparing predictions with existing Instream Flow Incremental Methodology predictions from the region).

There are several aspects of a river's flow regime which may influence its ability to maintain particular instream values. The current trend is towards methods that take a holistic view of the flow regime, of which consideration of adequate minimum flows are an integral part, along with consideration of some degree of flow variability to maintain the natural morphology and ecosystem. This more holistic approach aimed at defining 'environmental flows' becomes more important as the scale of abstraction increases, and suitable allocation limits or flow sharing rules are required in order to preserve flow variability across a range of scales.

A large number of methods have been used to determine flow requirements and "new" methods continue to be suggested. Instream flow methods can be conveniently divided into three types: historic flow, hydraulic, and habitat methods (which have recently been extended to include models that predict effects on fish directly, rather than on habitat). Another recent step has been the development of generalised habitat models, which combine the benefits of lower data requirements, associated with hydraulic methods, with the more explicit biological rationale of habitat methods.

In terms of minimum flow setting, research indicates that the mean annual low flow (MALF) is ecologically relevant to trout carrying capacity, because the MALF determines the average annual minimum living space for adult trout, and that it may be similarly relevant to native fish species, at least where habitat availability decreases with flow toward the MALF. This understanding can be applied to the setting of minimum flows in conjunction with the results of habitat modelling. If predicted habitat availability optima should occur at flows above the MALF, then habitat availability will be limited by the flow level at the MALF. In this case, flow decisions should be made to preserve a proportion of the habitat available at the MALF, with the aim of conserving instream values while still providing for 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 that optimum. The level of habitat retention is arbitrary, and it has been suggested that this level could be varied according to the relative value of instream resources.

This approach has already been applied by Horizons to setting minimum flows in their jurisdiction based on the output of IFIM habitat analyses. We found that the new generalised habitat models mostly produced similar minimum flow recommendations for streams in the Horizons region as those based on full IFIM habitat modelling.

Flow decisions should be science-based, but the effort put into the science ought to reflect the values of the instream resources. The values need to be weighed against the risk, and consequences of error in predictions based on the science.

Based on this premise we suggest a tiered approach to instream flow assessment and minimum flow setting. This approach consists of four methods that can be employed depending on the level of demand for water abstraction and the significance of instream values, these are:

1. Historical flow methods, where the minimum flow can be set according to historical flow statistics (e.g. the MALF or a proportion of it) if the total abstraction demand is a small proportion of river flow (e.g. <10% of the mean annual low flow, MALF) at any downstream point in the catchment;
2. Application of generalised habitat models, requiring a minimum of site investigation in cases where the total abstraction demand is moderate (e.g. <30% of MALF), or where the instream values are low;
3. Detailed site instream habitat analysis (e.g. IFIM) and consideration of effects where abstraction demand is high (e.g. >30% of MALF) and where the instream values are high;
4. The use of WAIORA to set flow requirements for small streams dominated by macrophytes, where dissolved oxygen concentration is a limiting factor. Note that this may have to be combined with other technical methods, for example groundwater modelling if drying of spring-fed streams is perceived as an issue.

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.

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1. INTRODUCTION

This report addresses options available to Horizons Regional Council for setting defensible minimum flows for rivers in its jurisdiction. Horizons is currently considering water allocation options for the various water management zones within its region, and an appropriate minimum flow is recognised as a key part of the water allocation framework. However, minimum flow setting is often subject to much debate in policy and resource consent decision making. At present Horizons have minimum flows set in regional plans for some of its rivers, but aims to incorporate specified minimum flows in the new regional plan ("The One Plan") to enable debate around minimum flows to occur at the policy level, as opposed to a consent-by-consent basis.

Like many other regional councils, Horizons have used the Instream Flow Incremental Methodology (IFIM) habitat modelling to inform minimum flow setting. However, this methodology is relatively expensive, time consuming, and relies on extensive and carefully timed field work during low flow periods. Horizons have an established programme of IFIM surveys, but are limited to completing two of these surveys in any one financial year. Consequently, Horizons have sought advice on alternative methods to inform interim minimum flow setting.

The generalised habitat models recently developed by Lamouroux & Jowett (2005), based on existing IFIM habitat assessment data sets from New Zealand, have been identified as a promising approach. This method was introduced to key stakeholders at a recent workshop held by Horizons. There appeared to be a consensus that these generalised models would provide an expedient method for deriving minimum flows, provided that they could be demonstrated to perform comparably with traditional IFIM habitat modelling undertaken in the region.

This report was commissioned to:

1. Summarise methods available for setting minimum flows.
2. Suggest which of these methods (or which combination of methods) could be applied to minimum flow setting within the Horizons region; including verification of the applicability of generalised habitat models to the Manawatu/Wanganui Region (by comparing predictions with existing IFIM predictions from the region).

2. THE FLOW REGIME AND ASSESSMENT FRAMEWORKS

There are several aspects of a river's flow regime which may influence its ability to maintain particular instream values. These include:

- Large floods, which are responsible for maintaining channel form and large scale sediment transport.
- Smaller floods and freshes, which flush fine sediment, periphyton and other aquatic vegetation.
- Low flows, the period of minimum wetted habitat availability, but also potentially of relatively high productivity in the remaining habitat.
- Flow recessions, higher than usual flow in the few days following a flood may offer enhanced recreational opportunity, and increased wetted area during flow recession over longer periods may enhance ecosystem productivity.
- Flow variability, at a range of scales. From seasonal variability comprising the annual flow regime, to small scale flow variations, which many people consider are an essential element of the regime that should be maintained, avoiding long periods of artificial "flat lining".

Long-term solutions to river flow management need to take a holistic view of the river system, including geology, fluvial morphological, sediment transport, riparian conditions, biological habitat and interactions, and water quality, both in a temporal and spatial sense.

The instream flow incremental methodology (IFIM; Bovee 1982) is an example of an interdisciplinary framework that can be used in a holistic way to determine an appropriate flow regime by considering the effects of flow changes on instream values, such as river morphology, physical habitat, water temperature, water quality, and sediment processes. Its use requires a high degree of knowledge about seasonal and life-stage requirements of species and inter-relationships of the various instream values or uses.

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

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

Historically, the focus of instream flow studies has been on determining the low flow conditions required to maintain particular instream values, because during low flows there is the greatest competition for the limited amount of water that is available, and the river ecosystem is most under stress. Instream flow methods have traditionally been used to define a minimum flow, below which no human influences should occur. At least in New Zealand this focus has been justifiable in the past because in most cases rates of abstraction have been relatively low, from run of the river type abstractions, with very little potential to influence any part of the flow regime other than the duration and magnitude of low flows. 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. This more holistic approach aimed at defining ‘environmental flows’ becomes more important as the scale of abstraction increases, and suitable allocation limits or flow sharing rules are required in order to preserve flow variability across a range of scales.

3. INSTREAM FLOW ASSESSMENT METHODS

A large number of methods have been used to determine flow requirements and “new” methods continue to be suggested, only a few of which are discussed here. The method or methods used to develop an appropriate minimum flow or flow regime will depend on the case being considered, and can vary from a quick rule-of-thumb assessment to detailed studies over several years. Even though methods have been applied for more than 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.

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.

3.1. 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.

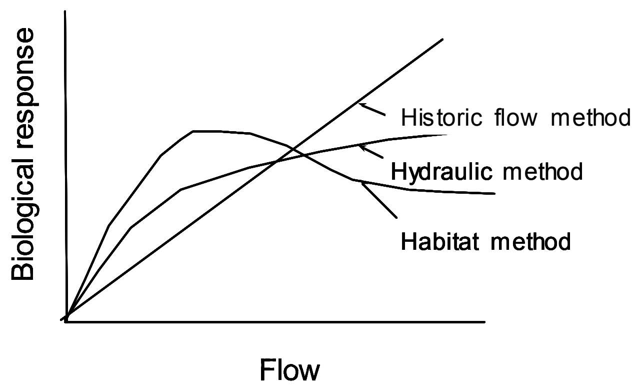


Figure 1. Hypothetical relationships between assumed biological response to flow for the historic flow, hydraulic and habitat methods. The biological response is assumed to be proportional to the flow, the wetted perimeter or width, and the weighted usable area, for the historic flow method, the hydraulic 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 m/s, 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 m/s and considered these to be in the good to optimum range for aquatic organisms. This is an example of a "regional method", applicable to 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 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.

3.2. 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.

3.3. Habitat methods

Habitat methods, including the habitat component of the Instream Flow Incremental Methodology (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.

3.3.1. Habitat models

Several one dimensional (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 (Appendix 1).

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.

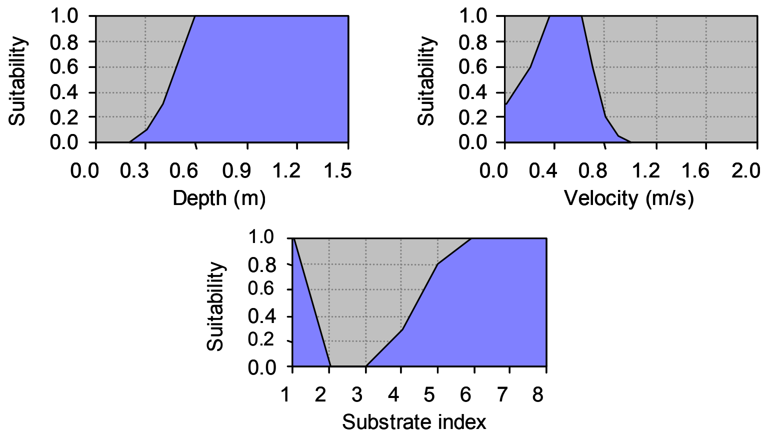


Figure 2. Habitat suitability curves for adult brown trout (adapted from Hayes & Jowett 1994).

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 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.

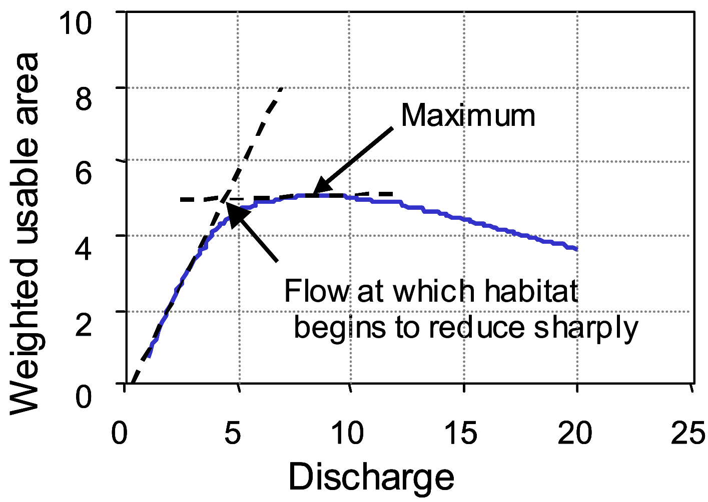


Figure 3. Selection of minimum flow at the breakpoint where habitat begins to decline sharply with decreasing flow.

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 such as kayaking, swimming and jet boating (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).

3.3.2. *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 (feeding on invertebrates drifting along in the water column) 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 on a spatially explicit basis at a fine scale), and carrying capacity (Hayes et al. 2000, 2003; Kelly et al. 2005).

3.4. Regional methods

Tennant's (1976) method is a good example of a regional method that combines the 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 m/s might be considered poor, 0.1-0.3 m/s adequate, and 0.3-0.5 m/s good for aquatic organisms such as trout and benthic invertebrates. Similarly, average depths greater than 0.15 m might be considered suitable for native fish and depths greater than 0.4 m suitable for adult trout.

These methods are potentially useful in that they combine the best features of habitat and flow methods 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.

3.5. 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.

More detail on generalised models, e.g. their derivation and data requirements, are given in Section 3.6.2 below. Generalised models can be implemented in WAIORA.

3.6. WAIORA – implements water quality and generalised habitat models

3.6.1. 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.

3.6.2. 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 in generalised habitat models (see Section 6).

4. MINIMUM FLOW ASSESSMENT: THE IMPORTANCE OF ECOLOGICALLY RELEVANT FLOW STATISTICS

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).

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 (Jowett & Hayes 2004). 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 for 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).

In contrast to long-lived species such as trout, some aquatic invertebrates have more than one cohort per year, and in New Zealand generally have asynchronous lifecycles (i.e. a range of different life stages are likely to be present at any given time), allowing them to rapidly repopulate areas following disturbance (e.g. by drift from tributaries and from other rivers by winged dispersal) (Williams & Hynes 1976; Scarsbrook 2000). Recolonisation of some river beds by benthic invertebrates following disturbance has been reported to occur within 4-10 weeks (Sagar 1983; Scrimgeour et al. 1988). 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 and median flow. 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 during periods in most years. Also in situations where optimum habitat occurs below the MALF, setting the minimum flow at MALF would result in less than optimum habitat availability. 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 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 stakeholders. 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.

5. INSTREAM FLOW ASSESSMENT FRAMEWORK

5.1. Overview

In their *Flow guidelines for instream values* reports MfE (1998) suggest a framework approach for assessing and managing flow dependent instream values. Figure 4 is an adaptation of that framework taking account of concepts suggested by Jowett & Hayes (2004) that assist with this approach. The adaptations are:

1. The identification of "critical values" as part of determining the management objective. 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). Candidates for critical value status might include flow sensitive rare or endangered species, or species with high fishery value. In cases where habitat analysis is used to inform flow management decisions, identifying critical flow dependent instream values circumvents the complexities of interpreting a range of different species' WUA curves independently.
2. Assigning the level of maintenance of the critical flow related factors that will ensure that the critical values identified in the management objective are maintained, according to the significance of these values. This concept is discussed more fully in Section 5.2 below. Note the distinction between *critical values*, which are what the flow management seeks to maintain (e.g. a population of a flow demanding endangered fish), and *critical factors*, which are the aspects of the river or flow regime that need to be managed to ensure the critical values are maintained (e.g. habitat for the endangered species of fish).
3. Considering the amount of water allocation before deciding on the appropriate technical assessment methods to be used. This step can save considerable effort being expended needlessly, where out-of-stream demand is low and is unlikely to increase substantially. In such cases the abstraction is unlikely to have a significant effect on the instream values and a conservative minimum flow (say, the natural MALF) is unlikely to impinge

significantly on security of supply for abstractors. Therefore, a low risk approach with minimal data requirements (setting a conservative minimum flow based on historical flow data, for example) may be justified, circumventing the need for expensive and time consuming instream flow assessment techniques. Consideration should be given to both current and probable future demand.

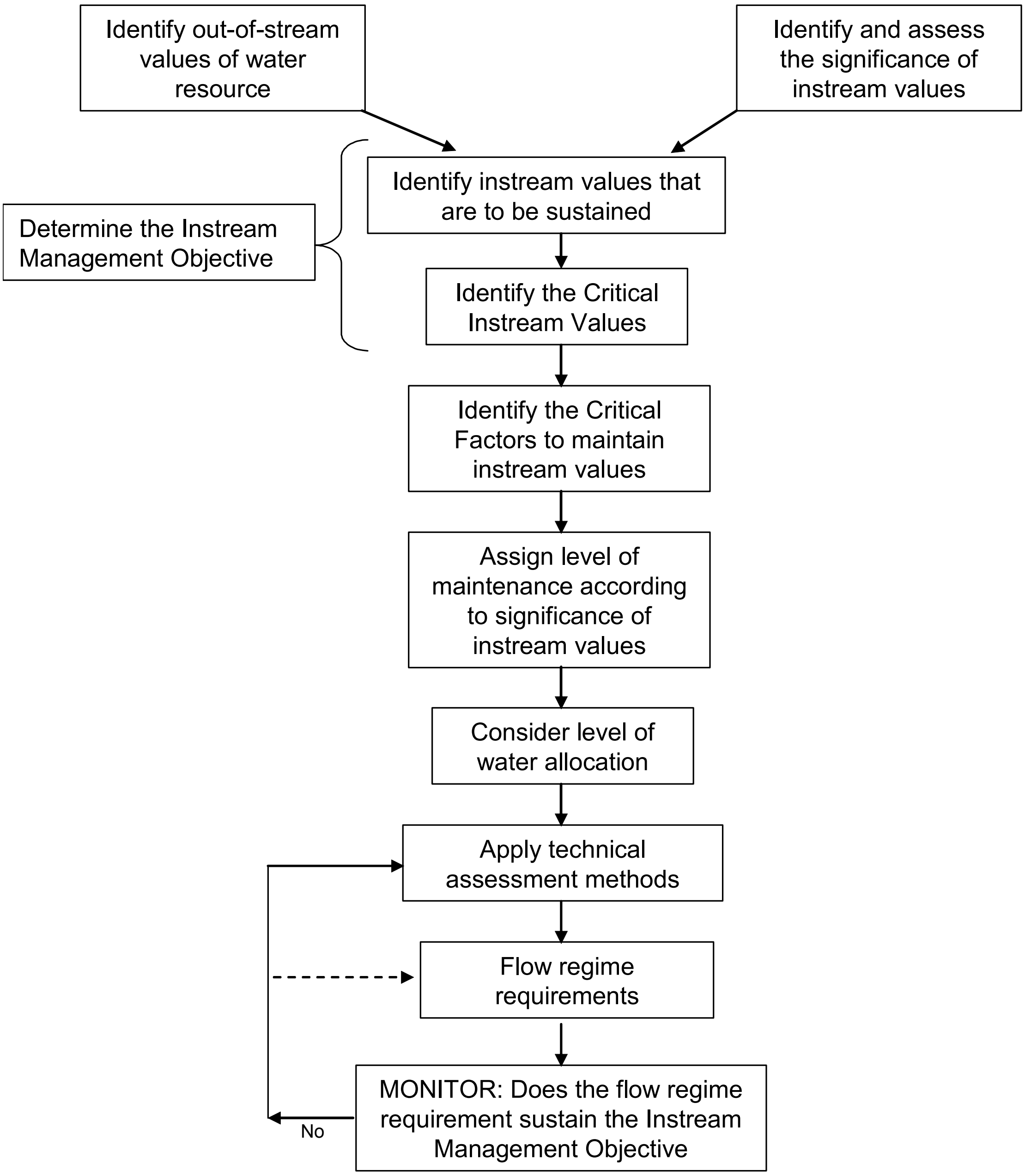


Figure 4. Framework for instream flow assessment, based on MfE (1998) Flow Guidelines for instream values, with adaptations from Hayes & Jowett (2004).

It is worth noting that monitoring following the implementation of flow regime management is likely to detect only large reductions in fish or invertebrate populations, given the high degree of natural variability in most New Zealand rivers. Even then the detection of change depends in large part on the existence of quantitative data documenting the state of the system over a reasonable duration prior to the implementation of flow management. However, this does not diminish the need to monitor the response of instream values to flow management.

5.2. Levels of habitat maintenance

In some situations deciding upon an ecologically defensible minimum flow, based on instream habitat modelling results, can be straight forward, such as when optimum habitat occurs below or at the ecologically relevant flow (Figure 5a). Traditionally the minimum flow is set either at the optimum (resulting in an increase in habitat) or at the breakpoint in the WUA x flow relationship. The breakpoint is the point of greatest rate of change in the WUA x flow curve – and can be justified on the basis that higher flows offer diminishing benefits for instream habitat, although there is no scientific evidence that the breakpoint is correlated with biological response.

Unfortunately, there are many situations where setting minimum flow is not nearly so straightforward. In small, or braided, rivers for instance, habitat, for flow demanding fish such as trout and torrentfish, optimum habitat occurs above the ecologically relevant flow and declines monotonically through the flow range under negotiation (Figure 5b). Therefore, there is no clearly identifiable point at which instream conditions become good or bad, but rather habitat simply gets worse as flow falls below the optimal value – although the rate of habitat change may vary with flow. In this case minimum flows necessarily have to be negotiated on the basis of incremental (or percentage) changes in habitat and decision making can be facilitated by deciding upon levels of habitat maintenance.

Levels of habitat maintenance are referenced against the habitat pertaining at the ecologically relevant flow statistics (e.g. MALF and median flow – see Section 4) and they are usually set arbitrarily. They are arbitrary partly because our state of knowledge on the effects of low flow is insufficient to predict how much instream values will change with a percentage flow reduction. Instream habitat modelling can estimate the incremental (or percentage) reduction in habitat as flow declines. This can assist stakeholder negotiation over minimum flows where it is useful to consider the relative values of instream versus out-of-stream values in the negotiation. However, how much habitat reduction is enough is more a matter of arbitrary stakeholder choice rather than ecological science.

In the absence of habitat modelling results, the assumption must be made that habitat is proportional to flow for flows less than the MALF. In this situation, a cautious approach to flow setting would maintain the amount of habitat provided by the MALF. Where habitat modelling results are available, we suggest that minimum flows be based on retaining a percentage of the amount of habitat at MALF for the critical value (or a proportion of maximum habitat if it occurs at a flow less than MALF). This approach has already been applied to recommending minimum flows to maintain instream values based on habitat modelling results in the Horizons region.

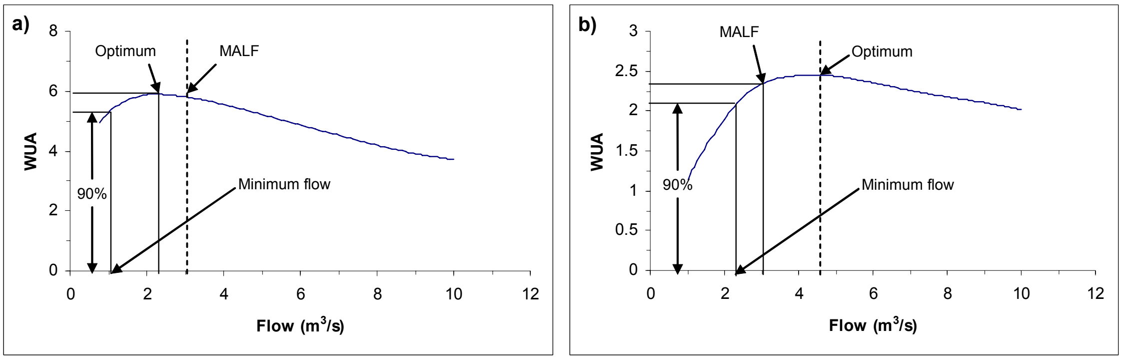


Figure 5. Derivation of minimum flow based on retention of a proportion (90% in this case) of available habitat (WUA) at a) the habitat optimum, or b) the MALF, whichever occurs at the lower flow, as recommended by Jowett & Hayes (2004).

The approach has been to vary the level of habitat retention according to the significance of instream values, based on the criteria suggested in Jowett & Hayes (2004) (Table 1). Ideally, the categories and levels of habitat retention ought to be set in consultation with the community and stakeholders. However, we believe the suggested levels of habitat retention in Table 1 are conservative, in that they are unlikely to be proportional to a population response. Theoretically, a change in available habitat will only result in a population change when all available habitat is in use (Orth 1987). In most cases, we believe that because flows are varying all the time, population densities are at less than maximum levels. That being the case, and speaking very broadly, a habitat retention level of, say 90%, would maintain existing population levels, whereas retention levels of 50% might result in some effect on populations, especially where densities were high.

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
Bullies e.g. upland, common, bluegill	-	5	60

In assessing the amount of habitat to be retained at low flow, it is important to realise that for some species, including many native fishes and juvenile trout, maximum habitat can occur at quite low flows. When the minimum flow is set at higher flows, as is the case when trout are the critical value, it will provide less than maximum habitat for the low flow species. We consider that the risk of detrimental effect of increasing the flow above that which provides maximum habitat for low flow species is not as great as decreasing the flow, and any habitat loss may be balanced by an increase in food production or the amount of cover. Low flow species will always have habitat available in the stream margins. The "best" brown trout rivers, such as the Mataura and Motueka, have flows that provide near maximum habitat between the mean annual low flow and the median flow.

Critical values and out-of-stream uses will need to be assessed on a catchment basis, because the significance of critical values will often increase as the river flow increases. The interim water allocation values currently being derived by Horizons in consultation with key stakeholders ought to assist with this. Small tributaries may have low significance ratings yet contribute to the flow of a river with high ratings. Maintenance of a minimum flow at the downstream site may depend on adequate flows in smaller tributaries. Flow requirements at points along the stream network need to be evaluated to identify the most downstream location with the highest flow demands. Ideally, this would be used as a monitoring site so that when flows at this site reach a minimum, water restrictions would be applied to all upstream consents. This is consistent with the approach already taken by Horizons (e.g. in the Upper Manawatu Allocation Plan).

6. GENERALISED HABITAT MODELS IN THE HORIZONS REGION

In this section we describe the derivation of generalised habitat models which predict habitat value (HV) for freshwater fish species found in streams in the Horizons region based on simple width – discharge relationships, as developed by Lamouroux & Jowett (2005). We then compare the performance of these models in deriving minimum flows with traditional habitat modelling in the Horizons region.

6.1. Derivation of the generalised method

The generalised habitat models adopted in this report are of the form described in Lamouroux & Jowett (2005) i.e.:

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

The values c and k describe the shape of the relationship between the dimensionless habitat value (HV) and discharge (Q , m^3/s) 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. A feature of this equation is that its peak occurs 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.

Model coefficients were derived for a range of taxa, based on two datasets. The first dataset included that used by Lamouroux & Jowett (2005), but with additional reaches held by Ian Jowett (NIWA). This dataset consisted of 175 surveyed reaches. However, in cases where there were several reaches from the same river, representing similar channel morphology, these were merged to give the most robust morphological representation of the river. This produced a total of 114 reaches to which the models were fit. Generalised models were also fit to 19 IFIM data sets provided by Horizons. Both datasets were composed mainly of typical single-channel gravel-bedded streams and rivers, although the NIWA dataset also included some braided river sections and some incised springfed streams. The reaches ranged from small tributary streams (e.g. Kiripaka Stream, a small tributary of the Waipa River near Whatawhata) to large rivers (e.g. the Clutha and Waitaki Rivers).

Lamouroux & Jowett (2005) fitted a non-linear mixed effects model, which described a common shape (a curve rising to a maximum and then declining) for each taxa (i.e. a common value of c and k were fitted across all reaches), but the scaling coefficient a was allowed to vary between reaches. However, this method cannot be used to fit generalised curves in some instances because the flow range modelled did not include the flow that provides maximum habitat for some reaches. For example, the optimum habitat for an edge-dwelling native fish species may not occur within the flow range that can be confidently modelled for a large river, given the constraints on extrapolating stage–discharge relationships to extremely high or low flows, relative to those on which they were developed. An alternative method of deriving generalised curves was used in this report to avoid the problem of modelling an inappropriate flow range. Instead of fitting one value of c and k to all reaches, values of c and k were fitted to each reach individually. Values for c and k were then examined and reaches with negative values and outlying values of c/k were excluded. Thus, the dataset for each species was reduced to those reaches that were able to provide optimum habitat availability for that species within the modelled flow range. The median values of c and k, taken over all the remaining reaches, were then adopted to provide a relatively unbiased representation of the shape of the habitat (HV) response to flow. Histograms of the peak of the HV curves (i.e. c/k) for the various reaches for a range of commonly habitat suitability criteria are presented in Figure 5.

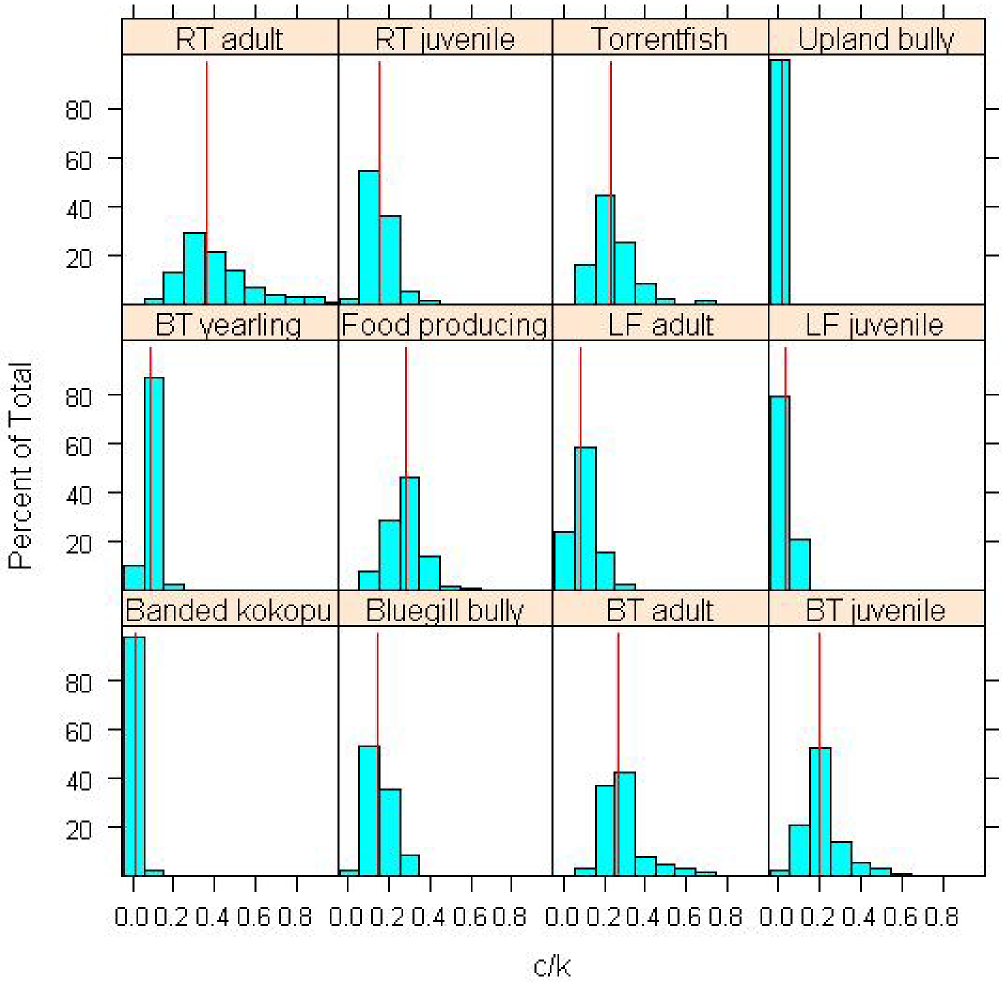


Figure 6. Histograms of c/k, i.e. specific discharge (discharge per metre of river width) at optimum HV for a range of species/ life stages, with the median values marked by the vertical red lines.

The median values of *c* and *k*, based on both NIWA's and Horizons' datasets are shown in Table 2 for a range of commonly applied habitat suitability criteria (HSC). The coefficient values from the Horizons dataset are reasonably similar to those from the NIWA dataset in most cases. The value of *c*/*k* is of particular relevance, because it defines the discharge per unit width at which the *HV* optimum occurs. The largest differences in *c*/*k* values between the NIWA and Horizons models were for food producing habitat (Waters 1976), adult rainbow trout (Thomas & Bovee 1993), and adult brown trout (Hayes & Jowett 1994). For these HSC the NIWA generalised models had a higher optimum, which would tend to produce more environmentally conservative minimum flow recommendations, if these were based on percent habitat retention.

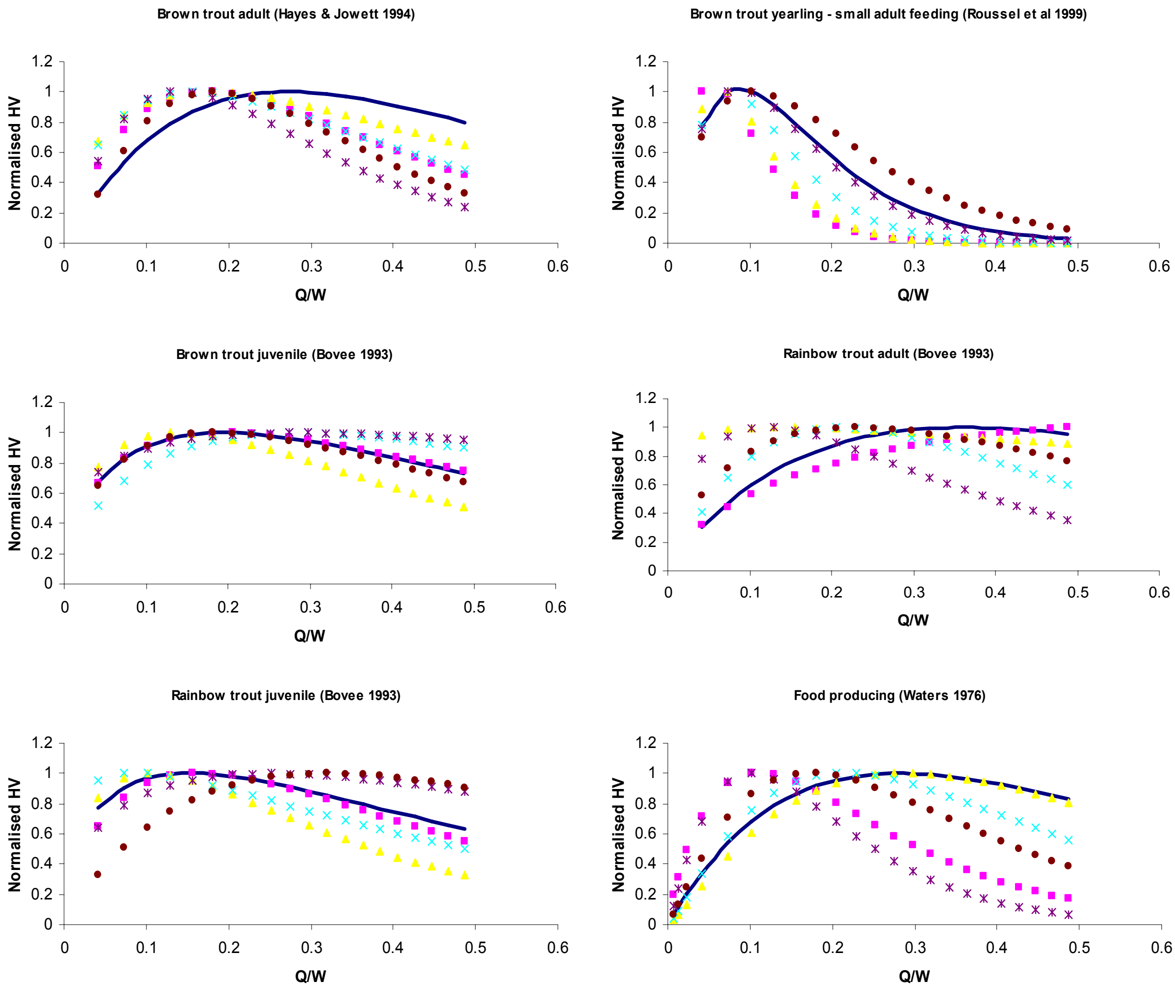
Since these generalised curves are based on the shape of the median *HV* response to specific discharge for each species, they ought to perform reasonably well for typical gravel bedded streams. However, they are not likely to model the response of *HV* to specific discharge well in streams toward the extremes of a channel form continuum, from deeply incised U-shaped channels (typical of spring-fed streams), to broad, relatively shallow unconstrained channels (typical of braided rivers). In the former average stream width changes relatively little with flow, so specific discharge increases relatively rapidly, while in the latter the opposite tends to be the case. We recommend that the generalised habitat models presented in this report are not applied to these types of channels. However, national or regional generalised models could be developed for these different river types in the future.

Table 2. Generalised habitat models used to predict habitat values (HV) from average characteristics of stream reaches. Model parameters *c* and *k* are developed for each reach and the median value selected, excluding reaches with negative values of *c* and *k* and outlying values of *c/k*.

Habitat suitability criteria	NIWA data			Horizons data		
	c	*k*	*c/k* [1]	*c*	*k*	*c/k* [1]
Brown trout adult (Hayes & Jowett 1994)	1.07	3.93	0.27	1.04	6.06	0.17
BT yearling (Rousel et al. 1999)	1.23	14.20	0.09	1.47	16.37	0.08
BT juvenile (Thomas & Bovee 1993)	0.53	2.70	0.19	0.51	2.68	0.19
Brown trout yearling (Raleigh et al. 1986)	0.36	3.96	0.09	0.28	4.62	0.06
Brown trout fry (Raleigh et al. 1986)	0.76	8.48	0.09	0.65	10.35	0.06
Brown trout spawning (Shirvell & Dungey 1983) [3]	1.16	10.92	0.11	1.04	13.76	0.08
Food producing habitat (Waters 1976)	1.01	3.59	0.28	1.22	7.07	0.17
RT adult (Thomas & Bovee 1993)	0.93	2.57	0.36	0.75	3.84	0.20
RT juvenile (Thomas & Bovee 1993)	0.45	2.92	0.15	0.45	3.42	0.13
Rainbow trout spawning (Jowett et al. 1996)	1.51	9.31	0.16	1.56	12.44	0.13
LF >300mm (Jellyman et al. 2003)	0.13	1.63	0.08	0.09	2.51	0.03
Longfin eel <300mm (Jowett & Richardson 1995)	0.07	2.24	0.03	0.11	1.84	0.06
Shortfin eel <300mm (Jowett & Richardson 1995)	0.16	2.64	0.06	0.32	4.00	0.08
Torrentfish (Jowett & Richardson 1995)	0.93	3.96	0.23	1.03	4.22	0.24
Shortjaw kokopu (McDowall et al. 1996) [2]	0.18	16.30	0.01	0.18	1.78	0.10
Banded kokopu (juvenile) (McCullough 1998)	0.16	12.91	0.01	0.26	31.43	0.01
Upland bully (Jowett & Richardson 1995)	0.11	8.82	0.01	0.20	11.60	0.02
Crans bully (Jowett & Richardson 1995)	0.09	6.78	0.01	0.14	8.48	0.02
Redfin bully (Jowett & Richardson 1995)	0.22	6.47	0.03	0.31	9.58	0.03
Bluegill bully (Jowett & Richardson 1995)	1.04	6.93	0.15	1.38	9.08	0.15

1 i.e. specific discharge (discharge per meter of river width) at HV optimum
2 suitability for cover locations only
3 Although Shirvell & Dungey (1983) measured water velocities 2 cm above the top of the redd, their velocities were within the range of mean water column velocities measured in North American rivers

Figures 6 and 7 depict *HV* predictions for a range of species in a hypothetical river using the median coefficient values (c, k) derived from the NIWA dataset, compared with the coefficients from five reaches chosen at random from the Horizons dataset. The scaling factor a has been excluded because for application to minimum flow setting it is superfluous; it controls only the magnitude of *HV*, whereas it is the shape of the curve that is most relevant to interpretation with regard to minimum flow setting. Note that *HV* has been normalised to one in each case (by dividing by the maximum value) to aid comparison. Using discharge per unit width allow different sized rivers to be compared on the same scale. These figures show that the median coefficient values, from the NIWA dataset, provide a reasonable approximation of the shape of the Horizons reach specific habitat values in most cases. For adult brown trout (Hayes & Jowett 1994), adult rainbow trout (Thomas & Bovee 1993), and food-producing habitat (Waters 1976) the NIWA coefficients produce higher optima than predicted by models from most of the Horizons reaches, as expected based on a comparison of median coefficients from both data sets (Table 2). However, as discussed above, the NIWA curves are likely to produce more conservative minimum flow recommendations as a result.



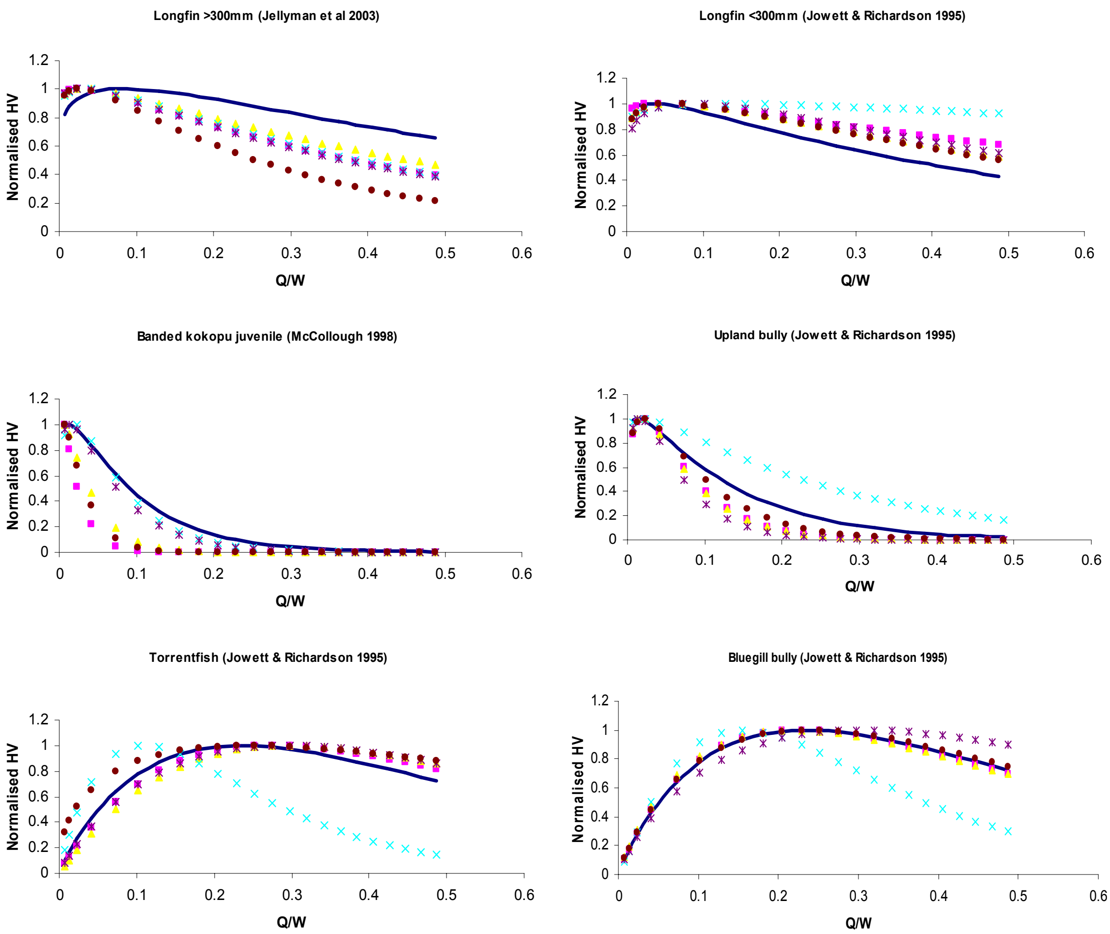


Figure 7. Comparison of normalised habitat per unit width (HV) predicted by the generalised habitat models using the median coefficient values from the NIWA dataset (solid blue line) and the coefficients for five reaches selected randomly from the Horizons dataset (various point markers), for a selection of trout and food producing (general benthic invertebrate) HSC.

The reasonable match in the shape of the *HV* response curve predicted by the median coefficients compared with those from randomly selected reaches is encouraging, because it suggests that the generalised model coefficient derived from a broad dataset of rivers around New Zealand are applicable to rivers in the Horizons region.

We used the median coefficient values from the NIWA dataset and the Horizons dataset to derive proposed minimum flows based on the habitat retention method outlined in Section 5.2. In most cases these proposed minimum flows compare well with those originally derived based on WUA output from RHYHABSIM (Table 3). We present results based on retention of a proportion of the predicted *HV* value (equivalent to HSI in RHYHABSIM, or WUA% in older versions), and of WUA (derived by multiplying *HV* values by river width; equivalent to WUA m^2/m in RHYHABSIM). The proposed minimum flows based on WUA m^2/m tend to be higher than those based on *HV* (Table 3). This is consistent with minimum flows based on the outputs of RHYHABSIM habitat modelling. WUA generally tends to peak at higher flows than HSI and consequently the minimum flows based on the former tend to be more environmentally conservative.

The median coefficients based on NIWA's dataset tend to produce higher proposed minimum flows, when these are based on Hayes & Jowett's (1994) adult brown trout, Shirvell & Dungey's (1983) brown trout spawning, or Thomas & Bovee's (1993) adult rainbow trout HSC (Table 3). Those based on retention of brown trout yearling - small adult feeding (Roussel et al. 1999) are mainly very similar between the NIWA and Horizons model predictions. However, there are a few instances where the Horizons model produces a higher minimum flow. Given that these models provide a coarser approximation of the likely response of habitat to flow than a full RHYHABSIM habitat analysis would provide, it would be prudent to take the cautious approach of adopting the generalised models that produce the most conservative minimum flows. Since these models are likely to be applied to minimum flow setting only where abstraction pressure is low to moderate, opting for environmentally conservative minimum flows should not be overly contentious.

The largest absolute disparity between minimum flows based on full IFIM habitat modelling and the generalised habitat curves occurred in the larger rivers (i.e. Manawatu at Hopelands and the Rangitikei River in all three survey reaches; but the original Rangitikei minimum flows are currently under review, see footnote to Table 1). The potential absolute disparity between minimum flows predicted by the different methods is likely to increase with river size, simply as a result of scaling (i.e. a similar percentage difference would translate to a larger absolute difference in larger rivers). This suggests that a conservative approach may be to restrict the application of generalised curves to minimum flow setting on relatively small streams. However, it is unlikely that the generalised habitat models would be applied to developing minimum flow recommendations for larger rivers, like the Manawatu at Hopelands and the Rangitikei, in any case. This is because both the instream values and demand for abstraction from such large rivers are likely to be significant enough to justify a full IFIM type habitat modelling exercise (see Section 7).

Table 3. Proposed minimum flows based on the generalised habitat models from the NIWA dataset and the Horizons dataset based on the habitat retention method compared with those originally derived based on full habitat modelling in RHYHABSIM. All minimum flows are based on retention of 90% of the habitat available at the MALF or habitat optimum, whichever occurs at the lower flow (except Oroua at Kawa Wool Site, which is based on 80% habitat retention). NB Table continued on next page.

Reach	MALF	Habitat suitability criteria on which minimum flow was based	Original minimum flow recommended	Equivalent minimum flow based on NIWA generalised curve HV	Equivalent minimum flow based on NIWA generalised curve WUA	Equivalent minimum flow based on Horizons generalised curve HV	Equivalent minimum flow based on Horizons generalised curve WUA
Manawatu at Hopelands Bridge	3.700	Brown trout adult (Hayes & Jowett 1994)	2.98	2.88	3.10	2.26	2.78
Manawatu at Weber Rd	1.875	Brown trout adult (Hayes & Jowett 1994)	1.60	1.58	1.62	1.46	1.52
Manawatu at Maunga Rd	1.113	Brown trout adult (Hayes & Jowett 1994)	0.97	0.94	0.98	0.90	0.95
Manawatu at Ormondville Takapau Rd	0.222	Brown trout yearling - small adult feeding (Roussel et al 1999)	0.20	0.19	0.20	0.19	0.20
During spawning		Brown trout spawning (Shirvell & Dungey 1983)	0.21	0.19	0.20	0.18	0.19
Manawatu at State Highway 2	0.14	Brown trout yearling - small adult feeding (Roussel et al 1999)	0.13	0.12	0.12	0.13	0.13
During spawning		Brown trout spawning (Shirvell & Dungey 1983)	0.13	0.12	0.12	0.12	0.12
Mangapapa Stm at Oxford Rd	0.03	Brown trout yearling - small adult feeding (Roussel et al 1999)	0.028	0.03	0.03	0.03	0.03
Raparapawai Stm at Gaisford Rd	0.080	Brown trout yearling - small adult feeding (Roussel et al 1999)	0.074	0.07	0.07	0.07	0.07
During spawning		Brown trout spawning (Shirvell & Dungey 1983)	0.075	0.07	0.07	0.07	0.07
Raparapawai Stm at Maharahara Rd	0.080	Brown trout yearling - small adult feeding (Roussel et al 1999)	0.074	0.07	0.07	0.07	0.07
Oruakeretaki Stm at State Highway 2	0.35	Brown trout yearling - small adult feeding (Roussel et al 1999)	0.293	0.28	0.30	0.29	0.30
Kumeti Stm at State Highway 2	0.070	Brown trout yearling - small adult feeding (Roussel et al 1999)	0.064	0.06	0.06	0.06	0.06
During spawning		Brown trout spawning (Shirvell & Dungey 1983)	0.070	0.06	0.06	0.06	0.06
Kumeti Stm at Te Rehunga	0.059	Brown trout yearling - small adult feeding (Roussel et al 1999)	0.055	0.05	0.05	0.05	0.05
Tamaki Rvr at State Highway 2	0.460	Brown trout yearling - small adult feeding (Roussel et al 1999)	0.360	0.38	0.39	0.39	0.40
During spawning		Brown trout spawning (Shirvell & Dungey 1983)	0.393	0.39	0.40	0.36	0.37
Tamaki Rvr at Water Supply Weir	0.260	Brown trout yearling - small adult feeding (Roussel et al 1999)	0.238	0.22	0.23	0.23	0.23
Mangatoro at Weber Rd	0.700	Brown trout yearling - small adult feeding (Roussel et al 1999)	0.33	0.32	0.38	0.35	0.41
During spawning		Brown trout spawning (Shirvell & Dungey 1983)	0.32	0.40	0.49	0.27	0.31
If management based on large brown trout habitat		Brown trout adult (Hayes & Jowett 1994)	0.59	0.57	0.61	0.53	0.58

Table 3. continued

Reach	MALF	Habitat suitability criteria that minimum flow was based on	Original minimum flow recommended	Equivalent minimum flow based on NIWA generalised curve HV	Equivalent minimum flow based on NIWA generalised curve WUA	Equivalent minimum flow based on Horizons generalised curve HV	Equivalent minimum flow based on Horizons generalised curve WUA
Oroua at Kawa Wool Site	1.2	Brown trout adult (Hayes & Jowett 1994)	0.95	0.85	0.90	0.76	0.82
Pohangina at Mais Reach	2.30	Brown trout adult (Hayes & Jowett 1994)	1.96	1.95	1.99	1.81	1.88
Rangitikei at Otara †	15.51	Rainbow trout adult (Thomas & Bovee 1993)	9.5	7.27	10.83	<5.0	8.67
Rangitikei at Otara †	13.14	Rainbow trout adult (Thomas & Bovee 1993)	8.75	6.95	10.10	<5.0	8.67
Rangitikei at Onepuhi †	17.93	Rainbow trout adult (Thomas & Bovee 1993)	14.55	8.70	10.45	<5.0	<5.0
Rangitikei at Onepuhi †	15.34	Rainbow trout adult (Thomas & Bovee 1993)	13.00	8.70	10.23	<5.0	<5.0
Rangitikei at Hamptons †	18.58	Rainbow trout adult (Thomas & Bovee 1993)	10.23	6.62	8.86	<5.0	<5.0
Rangitikei at Hamptons †	15.89	Rainbow trout adult (Thomas & Bovee 1993)	10.13	6.6	8.9	<5.0	<5.0

† The original minimum flow recommendations are currently under review. A bug discovered in RHYHABSIM during preparation of this report, appears to have resulted in inaccurate WUA predictions in the original analysis. This bug caused the physical habitat prediction to be sensitive to the order that the cross-sections were entered in the data file. The cross-section data for the Rangitikei reaches were entered in descending order rather then ascending order. New data checking routines in RHYHABSIM also indicated potential errors in the rating curves for some cross-sections in the Rangitikei reaches. A reanalysis of these reaches is now planned. Initial indications are that the reanalysis will produce minimum flow recommendations closer to those based on the NIWA generalised curve WUA in the Otara and Onepuhi reaches, but that the Hamptons Reach minimum flow will change little.

6.2. Data requirements for generalised habitat modelling

Aside from the generalised habitat model coefficients described above the only data requirement for applying these models is a width – discharge relationship for the study river.

There are at least three alternative methods of obtaining the wetted width – discharge relationship required to run the generalised habitat models. These are:

1. Direct field measurement
   - Measure wetted stream width across at least three locations in each habitat type (e.g. runs, riffles, pools) to cover the variation in width, for at least two known flows (as for WAIORA).

2. Remote sensing (e.g. LIDAR)
   - If remote sensing data are available with adequate resolution to measure average wetted stream width, and the flow at the time the data were collected is known, these measurements can be used in place of field measurements. Remotely sensed data could be used alone if at least two datasets at different flows are available, or a single dataset could be used in conjunction with a single episode of field measurement.

3. Use existing average hydraulic geometry relationships from the literature (e.g. Jowett 1998).
   - Jowett (1998) calculated an average river width (W) to flow (Q) relationship based on 73 New Zealand rivers of the form described above (i.e. $W = bQ^a$), with an exponent a of 0.176. The scaling constant b can be calculated from the equation and one measurement of average width at a known flow. (NB the exponent in this relationship ranged between 0.052 and 0.438 for individual rivers in the data set).

Methods 1 and 2 above (or a combination of them) are preferable because they provide a width – discharge relationship that is specific to the river in question. The coefficients of a river specific width – discharge relationship of the form:

$$W = bQ^a$$

can be derived where:

$$b = \frac{\log(W_1 / W_2)}{\log(Q_1 / Q_2)} \quad \text{and} \quad a = \frac{W_1}{Q_1^b}$$

Where W_1 and W_2 are the measured wetted widths at discharges Q_1 and Q_2, respectively (Jowett 1998).

7. RECOMMENDATIONS

The cost and effort of instream flow needs assessment ought to reflect the values of the instream resources. Ministry for the Environment's "Instream Flow Guidelines" (MfE 1998) suggest that the level of maintenance should reflect the merits of instream values in a particular river. Flow decisions should be science based, but the effort put into the science ought to reflect the values of the instream resources. The values need to be weighed against the risk, and consequences of error in predictions based on the science. For example, the consequences of error are much greater when the flow management aim is to protect a highly valued fishery, or a population of a rare species, than they are when the management aim is maintenance of a commonly occurring aquatic community that possesses only intrinsic value (Hayes 2004).

Based on this premise we suggest a tiered approach to instream flow assessment and minimum flow setting, as recommended by Jowett & Hayes (2004). This approach consists of four methods that can be employed depending on the level of demand for water abstraction and the significance of instream values, these are:

1. Historical flow methods, where the minimum flow can be set according to historical flow statistics (e.g. the MALF or a proportion of it) if the total abstraction demand is a small proportion of river flow (e.g. <10% of the mean annual low flow, MALF) at any downstream point in the catchment;
2. Application of generalised habitat models, requiring a minimum of site investigation in cases where the total abstraction demand is moderate (e.g. <30% of MALF), or where the instream values are low;
3. Detailed site instream habitat analysis (e.g. IFIM) and consideration of effects where abstraction demand is high (e.g. >30% of MALF) and where the instream values are high;
4. The use of WAIORA to set flow requirements for small streams dominated by macrophytes, where dissolved oxygen concentration is a limiting factor. Note that this may have to be combined with other technical methods, for example groundwater modelling if drying of spring-fed streams is perceived as an issue.

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. The approach to setting allocation limits, in conjunction with minimum flows, that Horizons has taken recently 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

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). We understand that the relationships between minimum flows, allocation limits and flow variability are to be the focus of further consultation associated with the drafting of the "One Plan", perhaps this concept could be considered further during this process.

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Ian Jowett and John Leathwick at NIWA developed the generalised habitat model coefficients based on both the NIWA and Horizons datasets. Jowett & Hayes' report to Environment Southland "Review of methods for setting water quantity conditions in the Environment Southland draft Regional Water Plan" provided much of the background for this report, including the review of instream flow assessment methods.

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10. APPENDICES

Appendix 1. Pros and cons of 1D and 2D hydraulic/habitat models

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.