Ministry for the Environment
Manatū Mō Te Taiao

Draft Guidelines for the Selection of Methods to Determine Ecological Flows and Water Levels

Prepared by
Beca Infrastructure Ltd

New Zealand Government

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Contents

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1. **Ian Jowett** (National Institute of Water and Atmosphere, NIWA) and **John Hayes** (Cawthron Institute) wrote Section 2 (Rivers).
2. **Clive Howard-Williams** and **Brian Sorrell** (NIWA) wrote Section 3 (Lakes and Wetlands).
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Executive Summary – Recommendations

Introduction

The Ministry for the Environment (MfE) is assessing the need for a National Environmental Standard (NES) on methods for establishing ecological flows and water levels for rivers, lakes, wetlands, and groundwater resources. As a part of this process, MfE sought scientific guidelines for selecting appropriate methods for determining ecological flows and water levels. Beca Infrastructure Ltd (Beca) was commissioned to coordinate the 'capture' of this advice from some of New Zealand's top experts on the science of assessing the ecological requirements for ecological flows and water levels. This executive summary documents which approach the expert group recommends to be taken in selecting an appropriate method. The full report provides the underlying logic behind the recommendations.

It should be noted that this report relates only to method selection for establishing ecological flow requirements. **Ecological** flows are defined here as "the flows and water levels required in a waterbody to provide for the ecological integrity of the flora and fauna present within waterbodies and their margins". This report offers no guidance on the process of how to set **environmental** flows (defined as "the flows and water levels required in a waterbody to provide for a given set of values which are established through a regional plan or other statutory process") or the management implications of environmental flow decisions.

Methodology

Beca facilitated a two-day workshop in Christchurch on 19–20 December 2006. The workshop participants:

(i) listed the ecological management objectives/values relating to the ecological flow/level of the river, lake, wetland or groundwater resource being considered, together with factors that might affect the ability to achieve that objective

(ii) listed the technical methods applicable to the setting of ecological flows and water levels for the type of water body under consideration and debated the pros and cons of each method

(iii) developed a matrix of methods applicable depending on the significance of the values perceived for the water resource under consideration, and the degree of hydrological alteration being considered for that water resource.

Subsequent to the workshop, lead writers – for each of: rivers, lakes and wetlands, and groundwaters – drafted documents intended to support the recommendations. Each of these documents was reviewed by three members of the workshop team as well as by the Department of Conservation (in the case of rivers and lakes) before being consolidated by Beca.

Recommendations: Rivers

- It is proposed that the approach to selecting technical methods to determine the ecosystem flow requirements of rivers be based initially on the risk of deleterious effects on instream habitat according to the species present and natural mean stream flow (Table 1). The risk of abstraction decreasing available habitat depends on stream size and the species present in the stream, with higher risks of deleterious effects in small streams than in larger streams and rivers.

Table 1: Assessment of risk of deleterious effects on instream habitat according to fish species present and natural mean stream flow (and generic application to other values/management objectives°). The data in the column for 'Salmonid spawning and rearing, torrentfish, bluegill bully', may be generically applied to invertebrates and riverine bird feeding (eg, wading birds, blue duck, black fronted tern).

Mean flow (m^3/s)	Inanga*, upland bully, Crans bully, banded kopopu*	Roundhead galaxias, flathead galaxias, lowland longjaw galaxias, redfin bully*, common bully*	Salmonid spawning and rearing, torrentfish*, bluegill bully*	Adult trout+
<0.25	High	High	High	High
< 0.75	Moderate	High	High	High
< 5.0	Low	Moderate	High	High
< 15.0	Low	Low	Moderate	High
15–20	Low	Low	Low	Moderate
> 20	Low	Low	Low	Low

* Access to and from the sea is necessary
+ Access to spawning and rearing areas is necessary
° Actual degree of impact will depend on the degree of hydrological alteration whether or not the level of risk is high or low

- The extent to which abstraction affects the duration of low flows is a useful measure of the degree of hydrological alteration. A high degree of hydrological alteration is assumed to occur when abstraction increases the duration of low-flow conditions to 30 days or more, with moderate and low levels of hydrological alteration corresponding to increases of about 20 days and 10 days, respectively.

The degree of hydrological alteration for a river can be determined, first by determining the risk based on mean flow and species present (Table 1), then using Table 2 to determine how the total abstraction (in terms of mean annual low flow, MALF) affects the degree of hydrological alteration for the stream and its risk category and its baseflow characteristics. In Table 2, a high baseflow river is one where the low flows are relatively high compared to the mean flow, such as in rivers with frequent freshes, rivers with their sources in hilly or mountainous areas or rivers fed from lakes, or springs. A low baseflow river is one where the low flows are very much lower than the mean flow, such as occurs in rain-fed rivers in areas that are not subject to orographic rainfall. Further details are given in the supporting document.

Table 2: Relationship between degree of hydrological alteration and total abstraction expressed as % of mean annual low flow for various risk classifications (Table 1) based on stream size and species composition.

Risk of deleterious effect						Degree* of hydrological alteration
Low risk and high baseflow	Low risk and low baseflow	Moderate risk and high baseflow	Moderate risk and low baseflow	High risk and high baseflow	High risk and low baseflow	
<20%	<15%	<15%	<10%	<15%	<10%	Low
20–40%	15–30%	15–30%	10–25%	15–30%	10–20%	Medium
>40%	>30%	>30%	>25%	> 30%	>20%	High

*Abstraction of more than 40% of MALF, or any flow alteration using impoundments would be considered a high degree of hydrological alteration, irrespective of region or source of flow.

- Once the degree of hydrological alteration is determined, Table 3 lists the technical methods that should be used to assess ecological flow requirements. One or more of the methods listed within each cell of Table 3 should be used to assess ecological flow requirements for the given combination of degrees of hydrological alteration and significance of instream values. In situations with high instream values, two or more methods from each cell should be used, because the risks to stream ecology of making an incorrect ecological flow decision are greater. The methods within each cell are not listed in hierarchical order and the choice of method(s) depends upon the perceived ecological problem affected by the flow regime. Specific recommendations of the use of each of the methods are given in the supporting document.

Hydrological alteration of rivers involves an examination of a number of hydrological statistics, including flow variability of the system, which affects the quality of instream habitat, and the connectivity of rivers with riparian wetlands, springs and groundwater. Potential critical factors include magnitude and duration of low flows or levels, timing, frequency and magnitude of floods and the inundation (as referenced to water level) of wetlands, surface–groundwater exchange, and maintenance of fish passage. This requires knowledge of the pattern and ecological significance of water level variation in wetland and groundwater systems.

Table 3: Methods used in the assessment of ecological flow requirements for degrees of hydrological alteration and significance of instream values.

Degree of hydrological alteration	Significance of instream values		
	Low	**Medium**	**High**
Low	Historical flow method Expert panel	Historical flow method Expert panel	Generalised habitat models 1D hydraulic habitat model Connectivity/fish passage Flow duration analysis
Medium	Historical flow method Expert panel Generalised habitat models	Generalised habitat models 1D hydraulic habitat model Connectivity/fish passage	1D hydraulic habitat model 2D hydraulic habitat model Dissolved oxygen model Temperature models Suspended sediment Fish bioenergetics model Groundwater model Seston flux Connectivity/fish passage Flow variability analysis
High	Generalised habitat models 1D Hydraulic habitat model Connectivity/fish passage Periphyton biomass model	Entrainment model 1D Hydraulic habitat model 2D Hydraulic habitat model Bank stability Dissolved oxygen model Temperature models Suspended sediment Fish bioenergetics model Inundation modelling Groundwater model Seston flux Connectivity/fish passage Periphyton biomass model	Entrainment model 1D Hydraulic habitat model 2D Hydraulic habitat model Bank stability Dissolved oxygen model Temperature models Suspended sediment Fish bioenergetics model Inundation modelling Groundwater model Seston flux Connectivity/fish passage Periphyton biomass model Flow variabiity analysis

Recommendations: Lakes and Wetlands

a. Lakes

The distribution and occurrence of healthy lake littoral habitats and communities vary with lake size, depth and water clarity. The risk of changing lake levels decreasing available habitat or adversely affecting communities depends on the lake bed profile (bathymetry), substrate type, water clarity, wave action as well as size and depth. The risks of deleterious effects are greater in shallower systems than in deep water bodies. Within a lake level range, impacts arise from changing seasonality in levels and the proportion of time spent at different levels (level duration).

- It is proposed that for lakes, the risks for a potential change to lake level may be defined as follows:
  - **Low**. Less than 0.5 m change to median lake level in lakes greater than 10 m depth, and less than 10% change in annual lake level fluctuation in lakes greater than 10 m depth; and less than 10% change in median lake level and annual lake level fluctuation in lakes less than 10 m depth; and, patterns of lake level seasonality (relative summer vs. winter levels) remain unchanged from the natural state.
  - **Medium**. Between 0.5 and 1.5 m change to median lake level and less than 20% change in annual lake level fluctuation in lakes greater than 10 m depth; and between 10 and 20% change in median lake level and annual lake level fluctuation in lakes less than 10 m depth; and, patterns of lake level seasonality (relative summer vs. winter levels) show a reverse from the natural state.
  - **High**. Greater than 1.5 m change to median lake level, and greater than 20% change in annual lake level fluctuation in lakes greater than 10 m depth, and more than 20% change in median lake level and annual lake level fluctuation in lakes less than 10 m depth; and, patterns of lake level seasonality (relative summer vs. winter levels) show a reverse from the natural state.
- The risks for a potential change to lake level must also be defined in relation to seasonal and inter-annual level variability as determined by the methods shown in Table 4 below and documented in full in the main report.
- Once the risk of potential change to lake level has been established (degree of hydrological alteration) the technical methods that should be used to assess level requirements should be selected from Table 4. One or more of the methods listed within each cell of Table 4 should be used to assess ecological flow and level requirements for the given combination of degrees of hydrological alteration and significance of instream values. In situations with high lake values, two or more methods from each cell should be used, because the risks to ecology of making an incorrect ecological flow decision are greater. The methods within each cell are not listed in hierarchical order and the choice of method(s) depends upon the perceived ecological problem affected by the flow regime. Specific recommendations of the use of each of the methods are given in the supporting document.

- The proposed categorisation of risks associated with potential changes in lake levels are based on the professional judgment/experience of lake experts within this team. We recommend that work be commissioned to provide scientific justification for this categorisation and provide an equivalent of MALF (and other flow statistics) based on level duration curves. Profiles of level duration demonstrate graphically and quantitatively the lake level regime, however there is currently no easy way to use these in a general rule-based format as they are calculated from absolute altitude. It will be possible to convert these to a relative level based on variance from a mean (or median) lake level. In this way curves between lakes could be compared and a general set of rules on level duration derived.

Table 4: Methods used in the assessment of ecological flow and water level requirements for degrees of hydrological alteration and significance of lake values.

Degree of hydrological alteration	**Lakes: Significance of values**		
	Low	***Medium***	***High***
Low	Historical time series analysis Expert panel	Historical time series analysis Expert panel	Habitat analysis in drawdown zone Water balance models Species-environment models Residence time vs. water quality modelling
Medium	Historical time series analysis Expert panel	Habitat analysis in drawdown zone Water balance models Species-environment models Residence time vs. water quality modelling	Bank stability and geomorphology analysis Wave action assessment Water level and ramping rates Water clarity assessments Temperature modelling Processes-based water quality models Groundwater/surface water interaction
High	Habitat analysis in drawdown zone Water balance models Species-environment models Residence time vs. water quality modelling	Bank stability and geomorphology analysis Wave action assessment Water level and ramping rates Water clarity assessments Temperature modelling Processes-based water quality models Groundwater/surface water interaction	Bank stability and geomorphology analysis Wave action assessment Water level and ramping rates Water clarity assessments Temperature modelling Processes-based water quality models Groundwater/surface water interaction Hydrodynamic water quality models

b. Wetlands

The distribution and occurrence of healthy wetlands varies with size and depth and connectivity to other hydrological systems. The risk of changing wetland levels decreasing available habitat or adversely affecting communities depends on the depth and the bathymetry and the dominant species present. Wetlands are generally shallow with wide littoral ephemeral areas that are dependent on a number of different flow-dependent variables. Therefore risks to wetlands are perhaps greatest compared with any other freshwater ecosystem. The risks of deleterious effects are greater in shallower than in deepwater wetlands, and wetlands without permanent connections to freshwater sources. The effect of changing inflows and/or outflows and therefore changing levels depends not only on the magnitude of change but also the timing, periodicity (hydroperiod) and duration of the levels.

- It is proposed that for wetlands the potential risk of ecological change associated with changes in levels may be defined as follows:
  - **Low**. Less than 0.2 m change in median water level; and, patterns of water level seasonality (summer vs. winter levels) remain unchanged from the natural state (summer relative to winter).
  - **Medium**. Greater than 0.2 m and less than 0.3 m change to median water level; and, patterns of water level seasonality show a reverse from the natural state (summer relative to winter).
  - **High**. Greater than 0.3 m change to median water level; and, patterns of water level seasonality show a reverse from the natural state (summer relative to winter).
- The risks for a potential change to wetland level must also be defined in relation to seasonal and inter-annual variability in hydroperiod as determined by the methods shown in Table 5 below and documented in full in the main report.
- Once the risk of potential change to wetland level has been established (degree of hydrological alteration) the technical methods that should be used to assess level requirements should be selected from Table 5. One or more of the methods listed within each cell of Table 5 should be used to assess ecological flow and level requirements for the given combination of degrees of hydrological alteration and significance of wetland values. In situations with high wetland value, two or more methods from each cell should be used, because the risks to ecology of making an incorrect ecological flow decision are greater. The methods within each cell are not listed in hierarchical order and the choice of method(s) depends upon the perceived ecological problem affected by the flow regime. Specific recommendations of the use of each of the methods are given in the supporting document.

Table 5: Methods used in the assessment of ecological flow and water level requirements for degrees of hydrological alteration and significance of wetland values.

Degree of hydrological alteration	Wetlands : Significance of values		
	Low	*Medium*	*High*
Low ***(< 20 cm change)***	Historical water level records Expert panel Remote delineation of site and catchment Wetland record sheet (MfE methodology)	Historical water level records Expert panel Remote delineation of site and catchment Wetland record sheet (MfE methodology)	Detailed local delineation Wetland hydrological condition assessment and model change (MfE methodology) Species-environment models Habitat assessment Water quality modelling
Medium ***(20–30 cm change)***	Historical water level records Expert panel Remote delineation of site and catchment Wetland record sheet (MfE methodology)	Detailed local delineation Wetland hydrological condition assessment and model change (MfE methodology) Species-environment models Habitat assessment Water quality modelling	Full ecohydrological assessment Groundwater /surface water interaction Process-based water quality models Microtopographic survey
High ***(> 30 cm change)***	Detailed local delineation Wetland hydrological condition assessment and model change (MfE methodology) Species-environment models Habitat assessment Water quality modelling	Full ecohydrological assessment Groundwater /surface water interaction Process-based water quality models Microtopographic survey	Full ecohydrological assessment Groundwater /surface water interaction Process-based water quality models Microtopographic survey

Recommendations: Groundwater

Typically, knowledge of groundwater systems is less certain than knowledge of surface waters. Therefore, the approach for groundwater differs slightly from the approach for rivers, lakes and wetlands. A 'cumulative approach' to groundwater methods application is used in response to uncertainty and the unknowns associated with groundwater systems. A 'cumulative approach' to methods application follows the typical groundwater investigation process whereby simple models are used to build more complex models.

- It is proposed that for groundwaters the potential risk for changes in levels may be defined as follows:
  - **Low**: Less than 10% of average annual recharge
  - **Medium**: 11% to 25% of average annual recharge
  - **High**: Greater than 26% of average annual recharge.
- Once the risk of potential change to groundwater levels has been established (degree of hydrological alteration) the technical methods that should be used to assess level requirements should be selected from Table 6. One or more of the methods listed within each cell of Table 6 should be used to assess ecological flow requirements for the given combination of degrees of hydrological alteration and significance of the

resource values. The methods within each cell are not listed in hierarchical order and the choice of method(s) depends upon the perceived ecological problem affected by the flow regime.Specific recommendations of the use of each of the methods are given in Chapter 4.

- Potential changes to flow regimes relate to the percentage allocation of aquifer recharge. It is acknowledged that these allocation thresholds from low to high may vary depending on the nature of the groundwater system. However the recharge percentages as presented, provide a conservative approach to groundwater allocation in most circumstances. 'Significance of values' should be used as the main criterion for determining methods most suitable for water level requirements when the relationship between groundwater allocation and the potential change to the flow regime is uncertain (eg, in deep confined aquifer systems where recharge and discharge are not well defined.

Table 6: Methods used in the assessment of water level requirements for degrees of hydrological alteration and significance of groundwater values.

Potential degree of hydrological alteration from groundwater allocation	Groundwater: Resource values and their relative significance		
	Low (not sensitive)	**Medium**	**High (extremely sensitive)**
Low (up to 10% of recharge)	Conceptual model /simple water balance Historical levels	Conceptual model /simple water balance Historical levels Expert panel Detailed water balance	Detailed water balance Time series analysis Analytical models Numerical quantity models – steady state Numerical quantity models – transient Numerical quality models – transport
Medium (11–25% of recharge)	Conceptual model / simple water balance Historical levels Expert panel	Detailed water balance Time series analysis Analytical models Numerical quantity models – steady state	Numerical quantity models – steady state Numerical quantity models – transient Numerical quality models – transport Consolidation models
High (over 25% of recharge)	Detailed water balance Time series analysis Analytical models Numerical quantity models – steady state Numerical quantity models – transient Numerical quality models – transport	Numerical quantity models – steady state Numerical quantity models – transient Numerical quality models – transport Consolidation models	Numerical quantity models – steady state Numerical quantity models – transient Numerical quality models – transport Consolidation models

1 Introduction

1.1 Purpose and Scope

This report provides scientific guidelines for the selection of methods to determine ecological flows and water levels for rivers, lakes, wetlands, and groundwater resources.

The process of establishing environmental flows and water levels is nested within wider environmental flow decisions. Environmental flows and water levels describe the water that remains in waterbodies to provide for ecological, tangata whenua, cultural, recreational, landscape and other values.

An *environmental* flow includes an 'ecological flow'. An *ecological* flow or water level is defined as:

> *the flows and water levels required to provide for the ecological integrity of the vegetation and fauna present within waterbodies and their margin.*

Therefore the ecological function of a waterbody must always be provided for when setting environmental flow management objectives; although other critical values may need to be taken into account in order to meet community expectations.

Environmental flow and water level decisions are made within the framework of the Resource Management Act (1991), national and regional policy statements, and the objectives and policies of the relevant regional plan.

In the setting of environmental flows there are two distinct elements:

- a robust scientific methodology for assessing the 'needs of freshwater ecosystems' over a range of flow and seasonal conditions
- a clear approach to taking into account the ecosystem values alongside other natural and development values of Māori and the wider community.

This report deals with the first of these elements and concentrates on ecological values. It does not take into consideration economic, social or cultural values, nor does it discuss RMA 'process' issues, which are beyond its scope.

1.2 The Context Around Use of Tables Within This Report

As described above, the process of establishing ecological flows is nested within wider environmental flow decisions. Ecological assessment should include the steps shown in blue in Figure 1.1, but these are nested within other components that are more related to process and qualitative decisions.

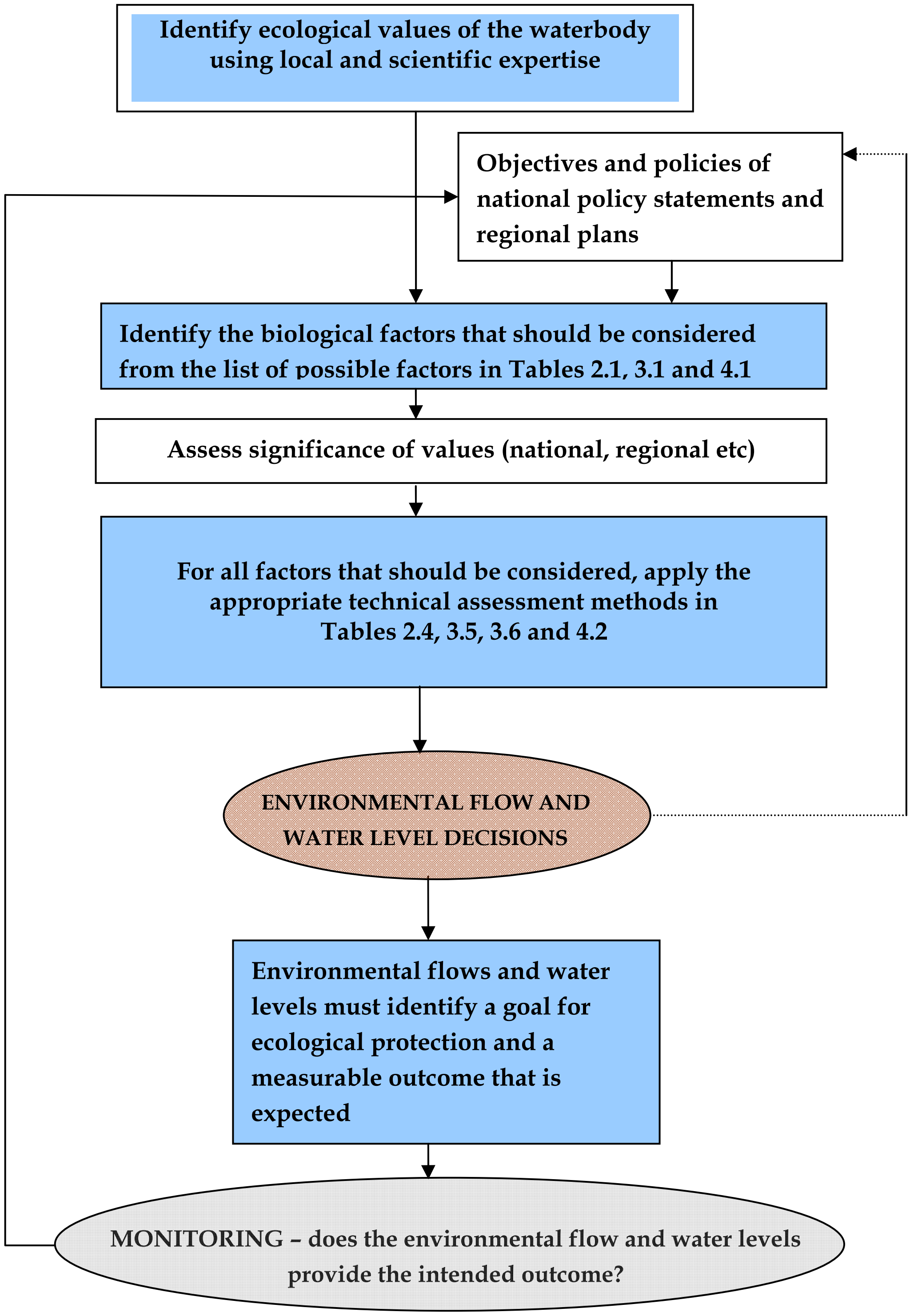

Figure 1.1: The process of assessing ecological flows within wider environmental flow decision-making.

The purpose of this report is to provide a framework for robust and scientific technical assessment methods. These should be applied to the assessment of hydrological requirements for freshwater biota in rivers, lakes and wetlands, and other important values in groundwater. The technical assessments assist the process of determining ecological flow and water level requirements.

1.3 Framework of the Approach

Traditionally, ecological flow methods have been used in rivers to define a minimum flow, below which there should be no human influence on river flow. However, taking into account the needs of all freshwater systems, the current trend is away from methods that set one minimum flow towards more holistic methods that consider the hydrological regime and aspects that, with some degree of hydrological variability, are needed to maintain the system morphology and ecologically-based values. Long-term solutions to flow and level management need to take a holistic view, taking into consideration geology, fluvial morphology, sediment transport, riparian conditions, biological community, habitat and interactions, connections between rivers, groundwater and wetlands, and water quality, both in a temporal and spatial sense. Similar holistic considerations apply to the social, economic and cultural aspects of environmental flows.

Technical methods need to be cost-effective and take a risk-based approach, with simple methods where the risk or environmental consequences of not achieving goals is low and more complex methods where aquatic values are high or the hydrological regime is highly modified.

When setting ecological flows, we need to acknowledge that the amount of hydrological variation required to maintain a healthy aquatic ecosystem is poorly understood. The complexity and unknown natural variation of many aquatic systems need to be acknowledged, and our relative lack of understanding of how different regimes will affect them. We need a cautious approach to setting flows that builds in buffers for risk and unknown outcomes. It is also important that no analytical method (eg, a model) should become a substitute for common sense, critical thinking about stream ecology, or careful evaluation of the consequences of flow modification.

An important component of ecological flows is that they quantify the amount the water available for allocation and also address requirements for both high and low flows/water levels throughout the year. The amount of water that can be allocated, and the manner in which it is used or regulated, determines the degree of hydrological alteration (the degree to which the natural hydrological regime could be potentially modified): this in turn will determine the technical methods to be used. For example, simple methods can be used to assess ecological flow and water level requirements where there are small changes to the natural hydrological regime. However, for substantial allocations or situations involving major flow/level regulation (eg, from impoundments), more complex and holistic methods are necessary. Similarly, the methods used will depend on the biota present or potentially present, so that there is no one method that fits all situations.

These draft guidelines:

i. provide a base methodology or process for establishing the relevant factors relating to a ecological value (for example, native fish)
ii. set an appropriate technical method or methods for describing how the value changes with changing hydrological patterns
iii. identifies methods and tools that are appropriate given the value and the level of allocation of a resource.

In summary, it aims to get a robust process for selecting methods that can be applied to assessing ecological flows and water levels.

The process involves:

i. assessing aquatic values and their relative significance
ii. determining the degree of hydrological alteration that could be expected from water allocation
iii. choosing an appropriate method for the assessment of ecological flow and water level requirements.

In the following sections, we describe this process applied to rivers, lakes and wetlands, and groundwaters, respectively. These descriptions provide the background and justification for our recommendations on the approach to selecting technical methods for the assessment ecological flows and water levels in freshwater systems under varying circumstances. The Executive Summary gives a brief overview of our recommendations.

1.4 Connected Systems

The document is structured into sections concerning with rivers, lakes/wetlands and groundwaters, each providing separate recommendations. However, it is well recognised that these systems are often inter-connected. For example, a groundwater ecological flow or water level may well be set to maintain flows in an adjacent river or in a spring. Similarly, flows in rivers can be set to ensure adequate ecological flows or water levels in wetlands along their margins or on their floodplains. For connected systems, it is recommended that the resource-specific tables are used in combination with each other; the most sensitive or significant value will drive then the selection of methods for all resources.

2 Rivers

2.1 Assessment of Instream Values and Critical Factors

An assessment of instream values is an important part of selecting an ecological flow regime for rivers. It establishes the biological communities, amenity and other values that could be affected and thus the methods of assessment to be used; it also establishes baseline data for the consideration of environmental effects. The identification of critical factors is also an important part of the process because instream values may be indirectly affected by flow-related factors that have a high flow requirement, such as provision of food and connectivity for some fish species.

Instream values may be grouped into:

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

There are, of course, overlaps and linkages among these values and in this report we focus on ecological or biological values, particularly 'flow-related values' that change in a discernible way as flow changes. Flow requirements of amenity values – such as fishing, boating, and swimming – can be determined using some of the same methods presented for biological values (eg, such as instream habitat models). Factors like water quality, water temperature and the micro-distribution of turbulence and velocity also change with flow; yet these flow-related changes are often small and the biological effects are difficult to predict because of the large natural variation in these factors and the tolerances of aquatic organisms. Table 2.1 lists the ecological (instream) values relevant to rivers and streams and possible factors that might affect the values and are considered to be flow-related to some degree. Some of the possible factors are closely related to flow, some of the factors apply only in special circumstances (eg, seston supply is specific to lake outlets), and in some instances the relationships between flow and the factors have not been established (eg, effects of flow on pH): further research is required to determine the relationship between the biological community, factor and flow requirements.

Table 2.1: Examples of biological instream values and management objectives.

Value/management objective	Possible factors that should be considered
Native fish and salmonids	Spawning habitat Rearing habitat Habitat of food sources Adult habitat / cover Access to spawning and rearing areas (including those in riparian wetlands and side channels) Passage of adults and juveniles Passage of predators or competitors Substrate Water quality and temperature
Invertebrates	Substrate Water depth and velocity Sediment transport / flow disturbance Water quality Periphyton Temperature Seston supply
Algae/macrophytes	Substrate, size, composition and stability Nutrients Water depth and velocity Invertebrate grazers pH Flow regime and sediment transport Temperature
Riparian and floodplain ecosystems	Frequency and duration of inundation Sediment supply
River birds	Habitat of food sources Nesting habitat Predators, predator-free islands Juvenile habitat Adult habitat Feeding habitat

Section 88 in the fourth schedule of the Resource Management Act (1991) states that an assessment of effects "shall be in such detail as corresponds with the scale and significance of the actual or potential effects…", and s92(4) RMA allows a consent authority to require further information from a consent applicant if necessary to "…better understand the nature of the activity…, the effect it will have on the environment, or the ways in which any adverse effects will be mitigated". In particular, the relevant biological matters are: s5(2)b "safeguarding the life-supporting capacity of air, water, soil and ecosystems", s6(c) "the protection of areas of significant indigenous vegetation and significant habitats of indigenous fauna", s7(d) "the intrinsic values of ecosystems", and s7(h) "the protection of the habitat of trout and salmon."

The level of biological assessment for fish, benthic invertebrate, periphyton and macrophyte communities will depend on the degree of hydrological alteration. Where the degree of alteration is small, adequate data may already be available. Where the degree of alteration is high, detailed and quantitative biological information is required to assess biological significance and establish a baseline for evaluation of potential effects.

2.1.1 Significance of Instream Values

The significance or relative importance of the instream values that are associated with a river informs the level of protection that should be considered, and also the technical methods used to assess ecological flow requirements. As the relative importance of instream values increases, the consequences of not meeting the environmental goals also increase. Because of this risk, the most robust and biologically supportable technical methods should be used to assess ecological flow requirements in highly valued rivers.

The significance of instream values can be judged by:

- national or regional significance of biological assemblage (biodiversity)
- species abundance, rarity or scarcity
- popularity (eg, for trout angling or whitebaiting).

A rigorous and consistent process should be used for determining national or regional significance. For example, it is very easy to say that a set of factors makes a river 'unique' because all rivers are different in some way. Most rivers are considered 'locally important' because they provide a source of food, water, valued biological community, or recreational activity and it is necessary to compare rivers over a reasonably large area to determine their relative importance.

The determination of significance is assisted by national databases, such as the Department of Conversation's electronic threatened species list (http://www.doc.govt.nz/templates/MultiPageDocumentTOC.aspx?id=39578), the New Zealand Freshwater Fish Database, and national surveys such as trout abundance (Teirney and Jowett 1990), Fish & Game Council trout angling surveys (eg, Unwin and Image 2003), and national inventories of rivers (eg, Teirney et al. 1982).

2.2 Determination of Degree of Hydrological Alteration

2.2.1 Components of Hydrological Alteration and Flow Variability

The biologically important components of a hydrological regime are:

- magnitude and duration of minimum flow
  For streams and rivers these set the limit to habitat quantity and can influence connectivity to other habitats such as wetlands.
- magnitude, frequency, and duration of high flows
  In streams/rivers it is the magnitude of high-flow events sufficient to cause substantial movement of fine particles (fine gravel or smaller) that is most relevant in this context. These occur moderately frequently and contribute to maintaining habitat 'quality' through flushing away accumulations of silt and periphyton from coarse sediments.

The magnitude of such flow perturbations is usually about 3–6 times the median flow (or 3–6 times the low flow in a highly regulated river) (Biggs and Close 1989; Clausen and Biggs 1997). While they cause sand and fine gravel movement, they seldom move larger bed sediments such as cobbles where invertebrates and fish hide during such events. Some designed 'flushing flow' events below storage dams may need to be higher in magnitude for removal of some periphyton and silts deeper in the gravels of regulated rivers.

- magnitude, frequency and duration of flood flow sufficient to cause substantial movement of the armour layer and erosion of banks in rivers
  These flows cause large disturbances to the river and its floodplain and often wash most periphyton, macrophytes and invertebrates from rivers, together with a large proportion of young introduced fish species (McIntosh 2000). Most native fish species appear to have evolved to cope with these floods and may take temporary refuge in more sheltered bank areas. Studies of New Zealand rivers indicate that flows of more than about ten times the mean flow or 40% of the mean annual maximum flow begin to move a substantial portion of the river bed (Clausen and Plew 2004).

The above discussion is focussed on hydrological alteration from the viewpoint of how instream habitat is affected. Consideration should also be given to whether hydrological alteration of rivers will affect connectivity of rivers with riparian wetlands, springs and groundwater. Potential critical factors include timing, frequency and magnitude of inundation (as referenced to water level) of wetlands, surface–groundwater exchange, and maintenance of fish passage. This requires knowledge of the pattern and ecological significance of water level variation in wetland and groundwater systems (see chapters 3 and 4).

The biologically important parts of a hydrological flow regime show that variability above the minimum flow is usually required to maintain healthy ecosystems. Flow variability is determined from the overall pattern of low and high flows during the year. Figure 2.1 shows the relative magnitude of the different high flow events during 2004/05 in the Waiau River, in southern New Zealand. Low flows that set habitat quantity are in the range of 70–100 m^3/s, whereas high flows that help maintain habitat quality are in the range of 300–600 m^3/s (ie, 3–6 times the low flows). Flood flows that can alter channel morphology and transport bed sediments are greater than about 1,000 m^3/s.

The biological effects of hydrological alteration will depend on source of flow, stream size, and biological community. The importance of low/minimum flows for the provision of adequate habitat quantity is fairly clear, as are the effects of large flood flows on channel structure and aquatic fauna. However, through extensive laboratory and field experimental studies we have now begun to understand the link between habitat quality and ecosystem productivity/health of New Zealand rivers, and how these are driven by the magnitude, frequency and duration of small floods/flushing flows.

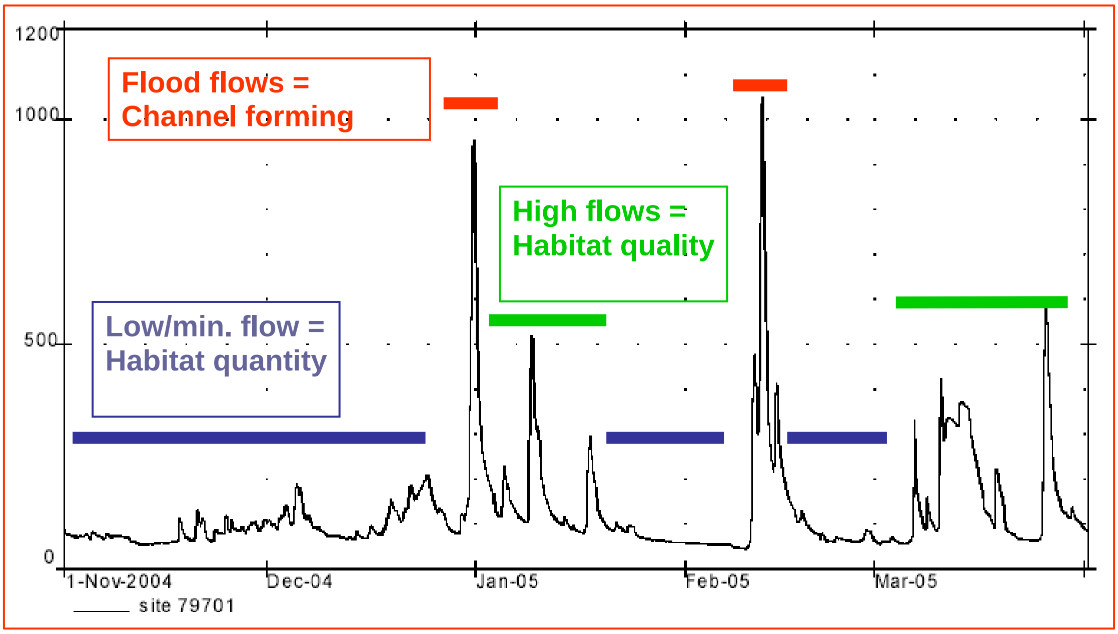

Figure 2.1: Different flow components in the Waiau River in southern New Zealand.

Lake outlets and spring-fed streams have few floods and freshes, but their biological communities are usually not considered degraded, with many spring-fed streams and lake outlets containing very high densities of trout and benthic invertebrates. In this type of river, natural low flows are usually relatively high for the channel shape (ie, the channel is relatively full), habitat quality is high, and sediment transport is low.

However, the source of flow in many streams is rainfall and this generates flows that are more variable than those from lake or groundwater sources. Rain-fed streams transport more fine sediment than spring- or lake-fed streams, so floods and freshes are necessary to remove fine sediment that accumulates during steady flows. In rain-fed streams, habitat quality in streams with frequent high flows tends to be higher than in streams with infrequent floods. This is simply because the magnitude of the low flows depends on the recession rate and time between high flows – the stream seldom has enough time between events to reach low flows.

Although the biological communities in streams with frequent floods and freshes are often considered to be more desirable than those in streams with infrequent floods, the biota reflect the quality of the habitat at low flow rather than flow variation. For instance, some lake outlet rivers, which have low flow variability and good water quality, have abundant populations of macroinvertebrate species which provide good trout food, and also provide good habitat for trout at low flow (Jowett 1992; Harding 1994). In contrast to lake and spring-fed rivers, long flow recessions that occur in rain-fed rivers with infrequent floods can result in poor-quality habitat, with aquatic communities that are associated with low-velocity environments (Jowett and Duncan 1990). Thus, stream biota will benefit from flow variation only in cases where habitat quality at low flow is poor and any flow variation increases the amount and quality of habitat, provided that there is sufficient time for a biological response.

The amount and quality of habitat at low flow varies with stream size and the flow recession rate and time between high flows, as described above. Habitat quality in small streams is often relatively poor at low flows and any further reduction in flow results in deterioration in habitat quality and consequent biological response. In comparison, flow reductions in large rivers will not necessarily result in a decline in habitat quality. Whether a river is considered small or large in this context depends on the biological community, because habitat and consequently flow requirements depend on the biological community that is to be supported.

Although flow variability is often thought of as an essential element of the flow regime that should be maintained, there is still relatively little evidence that flow variability affects instream communities either directly on indirectly in New Zealand rivers. Valued biological communities can be maintained in rivers where the flow regime has been extensively modified, but the needs of the instream values have been specifically identified and targeted in the management regime (Jowett and Biggs 2006) by some of the methods discussed here.

2.2.2 Degree of Hydrological Alteration and Flow Variability

The degree of hydrological alteration will depend on the way in which water is managed within the catchment. Water use can be divided into three categories of increasing hydrological alteration.

a. Consumptive use or abstraction

Water is taken from the river and used for activities such as water supply and irrigation, often with seasonally varying demand. The biologically relevant component affected is the magnitude of low flows, with a minor effect on duration. For example, abstraction of up to 10% of the mean annual low flow (MALF) is barely measurable and therefore unlikely to result in significant biological effects in any stream. Abstraction of up to 20% of MALF is unlikely to result in significant biological effects in lake- or spring-fed streams or in streams with frequent floods and freshes, such as those draining mountainous regions exposed to the prevailing westerly winds. When total abstraction exceeds these limits, the magnitude and duration of low flow may have significant effects on biota. In that case, measures such as a higher minimum flow (Dewson et al. 2007) or various flow-sharing regimes can be taken to avoid adverse biological effects.

b. Diversion or large scale abstraction

Water is diverted from the river on a relatively large scale and may be returned to the river downstream or discharged into another catchment. A diversion or abstraction is considered large-scale when it is able to divert more than 90% of the MALF out of a river. The biologically relevant components affected are the magnitude and duration of low flows. The frequency of flushing flows may also be affected if the capacity of the diversion is sufficiently large (eg, > 1.5 times the mean flow). With large-scale diversions or abstractions, the quality and amount of habitat at minimum flow will directly affect the biological communities because flows are at the minimum for substantial periods of time. Consequently, the minimum flow required to support these communities should be higher than the minimum flow that would be applied to situations with short-duration low flows (Dewson et al. 2007).

c. Storage regulation

River flows are modified by storage with potential change to the seasonality of flows, minimum flows, and high flows. Storage regulation can be consumptive (water supply or irrigation) or non-consumptive (hydro-electricity). The potential degree of regulation will depend on the storage volume in the impoundment. Storage regulation can affect all biologically important components of the flow regime.

2.2.3 Determination of the Degree of Hydrological Alteration

In order to develop a relationship between the potential abstraction (degree of hydrological alteration) and the effects on instream management objectives or values, one first needs to understand the risks involved. For management or preservation of fish communities, these risks are well understood and are related to stream size (decreasing available habitat) and the preferred flow requirements of the fish species present. For example, many small galaxids and bullies prefer low velocities and shallow water, juvenile salmonids tend to prefer moderate water velocities, and adult trout are commonly found in fast-flowing and deep water. Because water velocity and depth tend to increase with stream size, optimal stream size for various fish species can be broadly categorised (Table 2.2). The risks of deleterious effects on fish communities in small streams are higher than in larger streams and rivers. However, for other values such as invertebrates or river birds (Table 2.1), there is currently no strong quantitative relationship with flow. However, as explained in Section 2.1 (p. 5) the critical factors associated with fish community values can also be applied generically, which enable us to assess flow-related risk for some of these other values, based on their known relationship with fish communities.

In addition, flow in-river may well be managed (and therefore flows set) to provide flows to connected systems such as riparian wetlands, springs and groundwater flows.

The risks to fish communities (and where they can be applied generically to other biotic values) are categorised in Table 2.2 in terms of mean flow. Although other measures of flow (eg, median and mean annual low flow) could be used, mean flow is the measure that can be estimated with the greatest accuracy.

The extent to which abstraction affects the duration of low flows is a useful measure of the degree of hydrological alteration. Increasing the duration of low flows increases the risk of detrimental effects, and if low flows persist for 30–50 days per year, there will be noticeable growth of algae and changes to invertebrate communities and potential effects on fish (eg, Suren and Jowett 2006; Jowett et al. 2005).

A high degree of hydrological alteration is assumed to occur when abstraction increases the duration of low flow to about 30 days or more, with moderate and low levels of hydrological alteration corresponding to about 20 days and 10 days, respectively.

Table 2.2: Assessment of risk of deleterious effects on instream habitat according to fish species present and natural mean stream flow (and generic application to other values/management objectives)°. The data in the column for 'Salmonid spawning and rearing, torrentfish, bluegill bully', may be generically applied to invertebrates and riverine bird feeding (eg, wading birds, blue duck, black fronted tern).

Mean Flow (m^3/s)	Inanga*, upland bully, Crans bully, banded kopopu*	Roundhead galaxias, flathead galaxias, lowland longjaw galaxias, redfin bully*, common bully*	Salmonid spawning and rearing, torrentfish*, bluegill bully*	Adult trout+
<0.25	High	High	High	High
< 0.75	Moderate	High	High	High
< 5.0	Low	Moderate	High	High
< 15.0	Low	Low	Moderate	High
15–20	Low	Low	Low	Moderate
> 20	Low	Low	Low	Low

* Access to and from the sea is necessary
+ Access to spawning and rearing areas is necessary
° Actual degree of impact will depend on the degree of hydrological alteration whether or not the level of risk is high or low

In areas with frequent rain such as those exposed to the south or to the prevailing westerlies, flows are relatively reliable; and the mean annual low flow (usually defined as the average of the annual 7-day minimum flows) is a relatively high proportion of the mean flow and occurs for less than 3% of the time (11 days a year). Low flows are also relatively high compared to the mean flow in rivers fed from lakes or springs or pumice catchments. Abstraction of water increases the percentage of time flows are low, with the number of days per year at low flow (below MALF) increasing by 2–3 days for each 10% of MALF abstracted. In these rivers with reliable flows, abstraction of 40% of MALF will extend the duration of low flows to about 30 days.

In drier regions, such as eastern rivers with their source of flow in the rain shadow of ranges, flows are less reliable. The MALF is less than 1/20th of the mean flow and occurs for 6–10% of the time (c. 30 days per year). In these rivers, abstraction increases the duration of low flows by about 5–8 days per year for every 10% of MALF and abstraction of 20% of the MALF will extend the duration of low flows to about 40 days.

Table 2.3 lists the degree of hydrological alteration that would be caused by abstracting various amounts (10–40%) of the MALF for rivers with high and low baseflows for categories of risk based on stream size and species present. In this table, a high baseflow river is one where the mean flow is less than 20 times the MALF, such as occurs in rivers with frequent freshes, rivers with their sources in hilly or mountainous areas, or rivers fed from lakes or springs. A low baseflow river is one where the mean flow is more than 20 times the MALF. The degree of hydrological alteration for a river can be determined, first by determining the risk based on mean flow and fish species present (Table 2.2), then using Table 2.3 to determine how the total abstraction (in terms of MALF) affects the degree of hydrological alteration with the risk category and stream source of flow.

Table 2.3: Relationship between degree of hydrological alteration and total abstraction expressed as % of MALF for various risk classifications (Table 2.2) based on stream size and species composition.

Risk of deleterious effect						Degree of hydrological alteration
Low risk and high baseflow	Low risk and low baseflow	Moderate risk and high baseflow	Moderate risk and low baseflow	High risk and high baseflow	High risk and low baseflow	
<20%	<15%	<15%	<10%	<15%	<10%	Low
20–40%	15–30%	15–30%	10–25%	15–30%	10–20%	Medium
>40%	>30%	>30%	>25%	> 30%	>20%	High

Figure 2.2 shows the geographic distribution of rivers classified in the River Environment Classification (REC) as either high or low baseflow rivers based on the ratio of mean flow to mean annual low flow. Flow statistics in the REC are based on regional models and do not necessary account for spring sources; in some situations it may be necessary to assess baseflow status using local knowledge. Note that rivers with their sources in mountain areas can flow through areas containing rivers with low baseflows.

Once the degree of hydrological alteration is determined, appropriate technical methods can be selected to assess flow regime requirements (Section 2.3).

Abstraction of more than 40% of MALF, or any flow alteration using impoundments would be considered a high degree of hydrological alteration – irrespective of region or source of flow.

A discussion of the relationship between total allocation and the ecological flow requirements of rivers is given in Appendix 1.

2.3 Which Method? Decision-making Framework

The decision as to which method to apply for assessing the ecological flow requirements of a particular river depends firstly on the significance of the value to be managed and the critical factors affecting those values (Section 2.1). Gauging the significance of values requires stakeholder input (eg, Iwi, Department of Conservation, Fish & Game Council, interested general public). Identification of the critical factors affecting those values may also require expert input (eg, to assess whether water temperature is affecting or likely to affect fish or invertebrates).The decision on method depends secondly on the potential for hydrological alteration, which can be determined using the procedure outlined in Section 2.2. This framework (Table 2.4) allows selection of a methodology appropriate to the significance of the instream values and the potential for hydrological alteration. One or more of the methods listed within each cell of Table 2.4 should be used to assess ecological flow requirements for the given combination of degrees of hydrological alteration and significance of instream values. In situations with high instream values, two or more methods from each cell should be used, because the risks to stream ecology of making an incorrect ecological flow decision are greater.

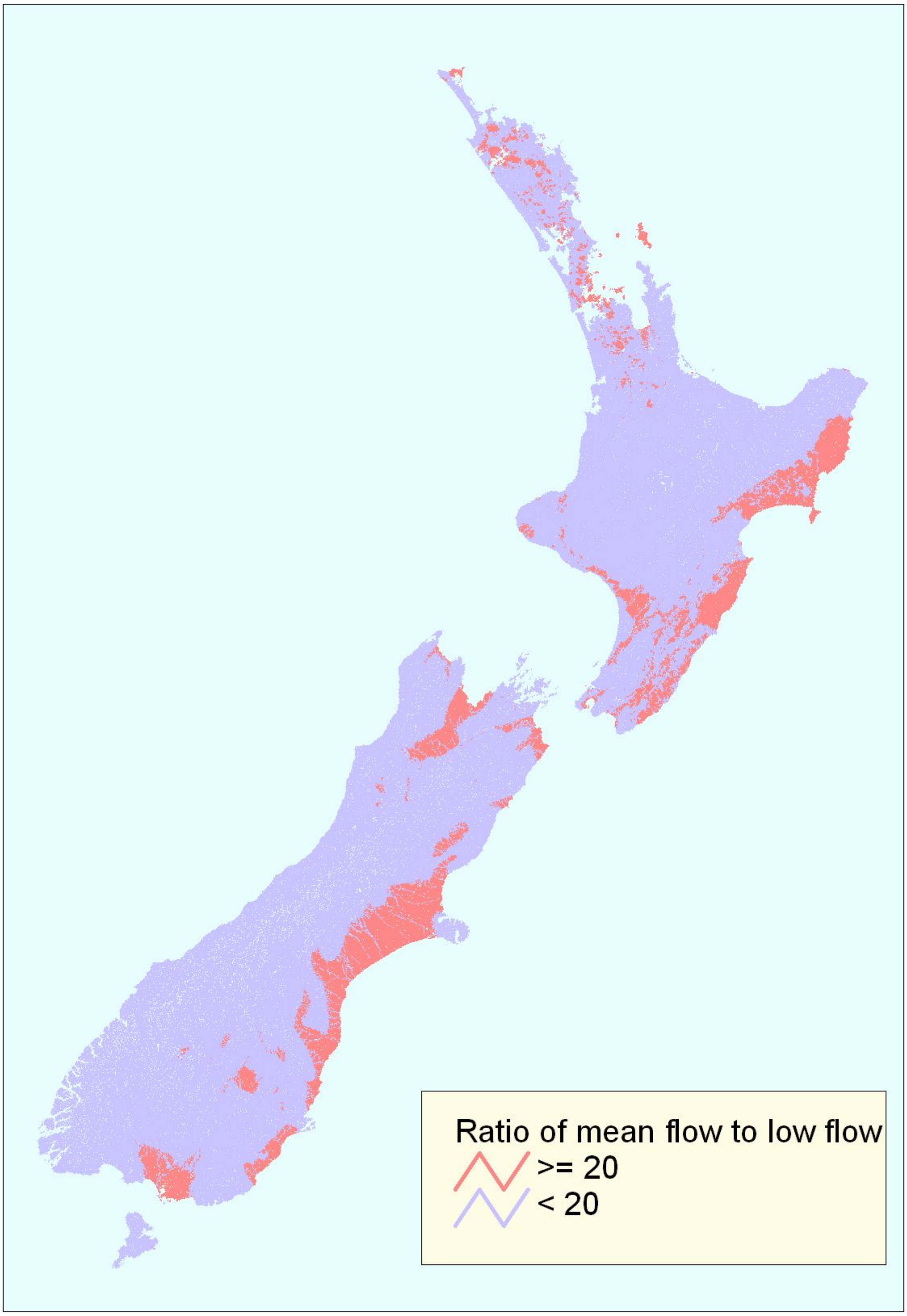

Figure 2.2: REC Classification of Baseflow Status, showing ratio of mean flow to low flow ≥ 20 (dark/red shading). and < 20(light/blue).

The methods within each cell in Table 2.4 are not listed in hierarchical order and the choice of method(s) depends upon the perceived ecological problem affected by the flow regime. For example, if elevated water temperatures affecting fish passage was the main concern under a medium alteration/high values scenario, then there would be little sense in using a hydraulic habitat model and vice versa.

Table 2.4: Methods used in the assessment of ecological flow requirements for degrees of hydrological alteration and significance of instream values.

Degree of hydrological alteration	Significance of instream values		
	Low	**Medium**	**High**
Low	Historical flow method Expert panel	Historical flow method Expert panel	Generalised habitat models 1D hydraulic habitat model Connectivity/fish passage Flow duration analysis
Medium	Historical flow method Expert panel Generalised habitat models	Generalised habitat models 1D hydraulic habitat model Connectivity/fish passage	1D hydraulic habitat model 2D hydraulic habitat model Dissolved oxygen model Temperature models Suspended sediment Fish bioenergetics model Groundwater model Seston flux Connectivity/fish passage Flow variability analysis
High	Generalised habitat models 1D Hydraulic habitat model Connectivity/fish passage Periphyton biomass model	Entrainment model 1D Hydraulic habitat model 2D Hydraulic habitat model Bank stability Dissolved oxygen model Temperature models Suspended sediment Fish bioenergetics model Inundation modelling Groundwater model Seston flux Connectivity/fish passage Periphyton biomass model	Entrainment model 1D Hydraulic habitat model 2D Hydraulic habitat model Bank stability Dissolved oxygen model Temperature models Suspended sediment Fish bioenergetics model Inundation modelling Groundwater model Seston flux Connectivity/fish passage Periphyton biomass model Flow variabiity analysis

The rationale behind this framework is that techniques that are simple to apply and require no or little additional data, are appropriate where both instream values and potential degree of hydrological alteration are low; but when either values or hydrological alteration increases, then more complex methods are justified. It can be seen from Table 2.4 that some methods listed are specific to establishing the ecological flows needed to manage critical factors (eg, temperature, seston flux, bank stability) whereas others are aimed more generally at maintaining aquatic habitat (eg, generalised habitat models, one-dimensional

(1D) or two-dimensional (2D) hydraulic habitat models). A summary of the methods listed in Table 2.4 is given in Section 2.4, including the advantages and disadvantages of each method.

2.4 Summary of Technical Methods

2.4.1 International Approaches

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

The approach to flow assessment currently favoured in Australia and South Africa is a ‘holistic’ approach that maintains a natural flow regime, low flows, seasonal variation, and flood frequency in order to protect aquatic fauna. A minimum-flow policy that restricts abstractions to the level of naturally occurring low flows and maintains major elements of the natural flow regime will maintain stream fauna, essentially in a natural state. This is a ‘safe’ environmental policy and one that will ensure the protection of aquatic resources in most situations. In this report, we suggest an approach that is cognisant of the holistic natural flow paradigm,while maintaining the biologically important elements of the flow regime.

Rivers will have different flow requirements depending upon the species that are supported by the river and their life cycle requirements. The challenge is to determine the aspects of the flow regime that are important for the various biota associated with their rivers, and to develop flow regimes that meet those needs. Experience in six New Zealand rivers has shown that flow regimes that are very different from the natural flow regime can sustain excellent fish and invertebrate populations and achieve instream management objectives (Jowett and Biggs 2006).

Annear et al. (2002) discuss instream assessment tools used in the United States and Canada and describe and comment on 29 different methods relating to hydrology, biology, geomorphology, water quality and connectivity. In the United States, the most commonly used method of assessing flow requirements is the instream flow incremental methodology (IFIM). This method is considered the most defensible method available at present and is particularly useful in ‘trade-off’ situations (Table 2.5). Low flow is not necessarily the factor that limits aquatic populations. Many studies have attempted to link stream fauna and its abundance to flow magnitude and most have failed to show any relationship. However, intuitively there must be a point at which there is too little water in a stream for the continued survival of aquatic species. This minimum flow is difficult to determine, and at present, instream habitat methods are the most biologically defensible approach to their determination.

An alternative 'standard setting' approach has been outlined by Richter et al. (2006) who use a range of variability approach (RVA) to derive a range of recommended flows for the low flows in each month, high flow pulses throughout the year, and floods with targeted inter-annual frequencies. The hydrological modeling behind RVA is a four-step process which characterises the streamflow record using 32 different hydrological parameters and the range in variation of these at plus or minus one standard deviation from the mean.

The latest iteration of the RVA method includes the addition of a number of new parameters, designed to deal with problems which had become apparent with the use of the method (these parameters were incorporated into the Indicators of Hydrological Alteration (IHA) software in 2005). These new parameters are grouped into five 'environmental flow components'(EFCs): extreme low flows, low flows, highflow pulses, small floods and large floods. The RVA method has been used the United States mainly in regulated systems to maximise the benefit of high-flow pulse releases of water from dams at a targeted magnitude, frequency, timing, duration and rate-of-change (Mathews and Richter 2007). To date the method has not been used in New Zealand for setting ecological flows and levels and is therefore not included in our recommended methods (Table 2.4). However, as the discussion document on the proposed National Environmental Standard for Ecological Flows makes clear, this technical document can be updated to reflect any new methods when their usefulness has been demonstrated in New Zealand. Further research is required on the relationship of RVA parameters to the biology, water quality, and geomorphology of river systems. Also the utility of the RVA method for setting ecological flows in New Zealand, particularly relating to abstraction, needs to be demonstrated.

Until the research discussed above is carried out, we propose that 'analysis of hydrological variation' should be included in the schedule of methods for rivers with a high significance of instream values. While analysis of hydrological variation will not by itself allow the setting of ecological flows, it will act as a 'flag' to other methods to illustrate the extent of hydrological change, and how these hydrological parameters may be affected by the ecological flow decision. Analysis of hydrological variation can be carried out using the RVA software or any other standard hydrological software that calculates flow statistics. Similarly, simple flow duration curves can be used where the proposed degree of hydrological alteration is low. Both analysis of flow variability and flow duration curves are standard hydrological techniques and are 'flags' to the potential importance of flow variability rather than ecological flow setting methods in their own right; therefore they are not discussed further in the description of individual methods (Section 2.5).

2.4.2 Habitat-based Methods in New Zealand – Uses and Criticisms

Minimum flow assessments based on hydraulic habitat have been used in New Zealand for 25 years. In that time there have been considerable improvements to the survey and analysis techniques, and to our knowledge of habitat preferences of New Zealand aquatic fauna and flora. The effectiveness of New Zealand flow assessments based on habitat methods has been examined and generally the response of aquatic communities has been consistent with habitat change predictions. Jowett and Biggs (2006) showed that an increase in minimum flow resulted in expected improvements for trout numbers in the Waiau River, but not in the Ohau River. Similarly, increases in minimum flow improved the benthic invertebrate communities in the Monowai and Moawhango rivers. They showed that a decrease in flow probably improved the trout fishery in the Tekapo River. Jowett et al. (2005) showed that low flows in the Waipara River were particularly detrimental to high velocity native fish species in the Waipara River. Richardson and Jowett (2006) showed that fish community changes in the Onekaka River were in accordance with habitat changes, with a decrease in minimum flow decreasing the abundance of koaro, but not redfin bullies. These are the only known case studies of how flow regime change has affected biological communities in New Zealand rivers, other than instances where rivers or streams have been completely dewatered with obvious consequences. One objective of the National Institute of Water and Atmosphere (NIWA)'s current research programme is to monitor and report the effects of flow regime changes so that we can learn from these experiences.

Habitat methods are based on hydraulic models that predict how water depths and velocities change with discharge. The same hydraulics models can also be used to evaluate the effects of flow regime changes on many aspects of the riverine environment, including sediment entrainment (for flushing flow and channel maintenance flow requirements), fish passage, water quality, sediment or seston deposition, and fish bioenergetics.

Habitat methods, although often described as micro-habitat, are in fact evaluating meso-habitat. The variation of flow with habitat can be determined from relatively few cross-sections and the selection of an appropriate tool will depend on the type of river and extent to which data are extrapolated. Habitat analyses based on simple hydraulic geometry, one-dimensional (1D) or two-dimensional (2D) surveys will often produce useful and similar results. The survey techniques described here are capable of predicting depths and velocities to the scale of the survey, which is usually measurements spaced at 0.1–3 m. This is consistent with habitat suitability observations that usually describe meso-habitats – the characteristics of the area in which the organism lives – rather than the micro-hydraulics of its precise location. In assessing suitability for one target species, we are often assessing conditions for a number of species that live in that area. Riffle-dwelling fish and invertebrates are an example, where the habitat suitability curves describe riffle conditions, rather than micro-habitat of the location of an individual organism. The selection of a target species (fish or invertebrate) as an indicator of stream health is a concept that can be applied to flow assessment.

The derivation and use of habitat suitability models are the most important aspects of flow evaluation. The tasks of survey, calibration, habitat suitability and analysis, and finally the interpretation of results require a good knowledge of river mechanics, hydraulics, and ecology. Survey (habitat mapping) and hydraulic calibration used in **r**iver **hy**draulic **hab**itat **sim**ulation (RHYHABSIM) are relatively robust; techniques such as water surface modelling and 2D modelling, are more complex.

Habitat suitability curves can be derived and used inappropriately. Although habitat suitability criteria are available for many New Zealand aquatic organisms, they can be improved by collecting more data and recalculated habitat suitability models. The question of hydraulic scaling, or transferability between rivers of markedly different size and gradient, for benthic invertebrate and rainbow trout habitat is a problem that has yet to be solved.

Although the functions of flow regime components (such as low flow, flow variability, flushing flows, and channel maintenance flows) are known, we do not know the degree to which the frequency and duration of these events affect biota; and we do not have any quantitative method of assigning acceptable frequencies and durations, other than mimicking nature. However, for periphyton and benthic invertebrates it is possible to provide rough guidance on an appropriate flushing flow frequency based on periphyton growth rates and reported invertebrate colonisation times.

Hudson et al. (2003) were critical of the application of IFIM and **phy**sical **hab**itat **sim**ulation/ river hydraulic habitat simulation (PHABSIM/RHYHABSIM) in resource management decisions in New Zealand. Their biggest concerns were the lack of knowledge around the habitat preferences of many of New Zealand's freshwater biota. As stated earlier, without good knowledge of these requirements it is difficult to make predictions about how changes in flow will affect available habitat for those species. Hudson et al. (2003) went so far as to say ".. habitat suitability curves have been developed for a very limited range of conditions." As noted earlier, habitat suitability curves are very important, and although we believe that the existing curves are based on hydraulic conditions commonly experienced in New Zealand rivers, we support additional collection of information. Native fish preference curves are being revised using a database of over 6,000 observations.

Hudson et al. (2003) were also concerned that the effects of other habitat characteristics, (such as water temperature and water quality) on freshwater biota that will also be affected by flow regimes, were rarely considered. However, IFIM involves considering all aspects of the instream environment that change with flow and this is recommended in this report for situations where there is a high degree of alteration.

Finally, hydraulic-habitat modelling is a tool to assist the decision-making process. No flow will maintain maximum habitat for all aquatic organisms. The selection of an appropriate flow regime for a river requires clear goals and target management objectives, with levels of protection set according to the relative values of the in- and out-of-stream resources. The process of establishing target management objectives is not a wish-list: management objectives should be relevant, important, flow-dependent and hierarchical. Failure to establish clear management goals and to carry out wide consultation will lead to conflict.

2.4.3 Summary of Technical Methods – Categories, Situations of Use, Pros and Cons

Annear et al. (2002) discuss instream flow assessment tools that are used in the United States. They divide them into three categories: standard setting where the method defines a flow; incremental where the method shows how stream characteristics vary incrementally with flow; and monitoring or diagnostic methods that assess conditions over time and compare the two broad categories of flow assessment (Table 2.5). The monitoring and diagnostic methods include indices of biotic integrity and hydrological alteration, and are not discussed in this report.

Table 2.5: Relative Attributes of the Two Broad Categories of Flow Assessment (from Annear et al. 2002).

Standard setting	Incremental
Low controversy	High controversy
Reconnaissance level planning	Project specific
Few decision variables	Many decision variables
Fast	Lengthy
Inexpensive	Expensive
Rule-of-thumb	In-depth knowledge required
Less scientifically accepted	More scientifically accepted
Not well-suited to bargaining	Designed for bargaining
Based on historical records	Based on fish or habitat

Although the more complex incremental methods based on hydraulic models could be used in every situation, it would not be cost-effective where values are low, nor would it be necessary to evaluate effects for aspects of the natural flow regime that would not be changed with proposed water allocation. Thus, the methods we suggest for assessing ecological flow requirements depend on the degree of hydrological alteration and the value of the instream resource that is affected (Table 2.4).

Technical methods of flow assessment that are currently available and are used in New Zealand are either standard setting methods (historical flows or expert panel assessments) or incremental methods (hydraulic-habitat). Traditionally, and appropriately, standard setting methods have been used to assess ecological flow requirements in situations where the natural flow regime is not expected to change markedly (except at low flows) and instream values are low to moderate. Hydraulic-habitat methods have tended to be used where the flow regimes are expected to change significantly, such as below a large-scale diversion or impoundment, or where instream values are high. Situations where each method might be used (related to type of hydrological alteration) are given in Table 2.6, and the pros and cons of each method are summarised in Table 2.7. A description of each individual method is given in Section 2.5.

Table 2.6: Methods used in the assessment of ecological flow requirements and the types of hydrological alteration for which they could be used.

Rivers and streams	Situation where model could be used	Tool
Historical flow methods	Abstraction of water	
Regional methods		
Expert panel		
Flow variability analysis	Assessment of the degree of hydrological alteration and identification of elements of flow regime that are changed	RVA (Range of Variability), Indices of Hydrological Alteration (IHA), Ecological Limits of Hydrological Alteration (ELOHA)
Generalised habitat models	Abstraction of water	WAIORA (Water Allocation Impacts on River Attributes)
Periphyton biomass model	Impoundments or diversions where the frequency of floods and freshes is altered	Look-up Table in Periphyton Guidelines
1D Hydraulic model and habitat evaluation	Abstractions, diversions, or impoundments	RHYHABSIM (river hydraulic habitat simulation), PHABSIM (physical habitat simulation), RHABSIM (river habitat simulation)
2D Hydraulic model and habitat evaluation	Abstractions, diversions, or impoundments affecting hydraulically complex rivers (eg, braided rivers)	River2D, Hydro2de (Hydro2de habitat analysis is done by post-processing of results)
Connectivity/fish passage assessment	Abstractions, diversions, or impoundments, where passage is thought to be an issue	1D models (2D models do not have passage analysis capability and analysis is done by post-processing of results either visually or numerically)
Entrainment model	Impoundments or diversions where the frequency of floods and freshes is altered	RHYHABSIM (1D model) and post-processing of 2D model results
Sediment transport		
Bank stability	Impoundments or diversions where the frequency of floods and freshes is altered	Post-processing of 1D and 2D model results
Suspended sediment	Significant abstractions, diversions, or impoundments, where suspended sediment or water clarity is an issue	RHYHABSIM (1D model)
Seston flux	Significant abstractions, diversions at or near below lake outlets	
Inundation model	Impoundments or diversions where the frequency of floods and freshes is altered	2D models
Flow variation analysis	Impoundments or diversions where the frequency of flow fluctuations is altered	1D models (RHYHABSIM or HABEF HABTAM in PHABSIM)
Fish bioenergetics model	Significant abstractions, diversions, or impoundments involving important trout fisheries	Kelly et al. (2005) (ie, Cawthron model), Addley (2006), Booker et al. (2004)
Water quality models	Abstractions, diversions, or impoundments in low-gradient macrophyte-dominated streams	WAIORA, RHYHABSIM
Groundwater models	Significant groundwater abstractions in alluvial river valleys	

Table 2.7: Pros and cons of flow assessment methods.

Rivers and streams	Description	Pros	Cons
Historical flow method 1	Proportion of recorded or estimated flows (eg, retain at least 90% of natural flow). Can be adjusted seasonally.	Quick and easy, uses existing data, results in flow variability without going to detailed level of analysis. Some abstraction allowed during times of low flow.	Assumes a linear relationship between flows and habitat, inconsistencies in estimating flow data, difficult to apply in un-gauged systems without accurate models, natural mistrust of method due to being too simple, doesn't target the needs of specific values. Not applicable where instream values are high or where is a large change to the natural flow regime.
Historical flow method 2	Minimum flow based on a proportion of a flow statistic, (eg, minimum flow 90% of 7-day mean annual low flow (MALF)). Can be adjusted seasonally.	Quick and easy, uses existing data. Widely used and well understood. Abstraction ceases when flows are less than minimum.	
Expert panel	Panel of experts to advise on flow requirements based on bankside inspection.	Quick, cheap, has credibility (dependent on experts), useful political tool to help overcome mistrust if well managed. Can be used to support other methods.	Not predictive, it is difficult for experts to determine how character of river changes with flow. Consensus can lead to poor ecological outcomes.
Range of variability approach (RVA)	Based on flows not exceeding the natural range of variation of a number of hydrological parameters that characterise minimum flows and flow variability. The RVA method recommends that flows be maintained within flow-management targets at the same frequency that would have occurred naturally. Flow-management targets can be within percentiles of the natural range or within standard deviations The RVA does not recommend maintaining flows exclusively at or near the level of the lower limit.	Allows interim flow targets and river management strategies to be developed where there is long-term hydrological data. Requires no field work or ecological evaluation. A conservative method suitable for rivers for which the objective is to keep them in their natural state.	Tends to be conservative in its allocation. More difficult to apply to systems with no historical flow records. Natural variation in parameters has not been related to biology, water quality or geomorphology and the method relies solely on maintaining the natural flow hydrograph. As the method stands, it is not clear how it could be applied to flow abstractions. May not be applicable where the sediment regime has also been altered. Not yet used in New Zealand for setting ecological flows.

Rivers and streams	Description	Pros	Cons
Generalised habitat models	Describes relationship between habitat and flow, simplified versions of detailed 1D habitat models.	Don't require full instream habitat surveys, could be used more widely.	Models lack information that could be gathered using a full 1D habitat survey; not as precise, relatively new technique, some restrictions to stream types they can be applied to (eg, braided rivers, spring fed streams).
Periphyton biomass model	Prediction of maximum biomass as a function of time between floods and nutrient concentrations.	Most relevant for rivers with major impoundments or large diversions where there is opportunity to manage flood flows.	Need accurate estimates of average soluble nutrient loads, needs more validation of accuracy of predictions.
1D Hydraulic model and habitat evaluation	Predicts water depth, velocity, and habitat suitability as a function of flow.	Widely used and understood, relatively easy modelling, gives a specific relationship, most closely links hydraulic habitat availability with a range of flows. Cooperative (multi-agency) studies can lead to better buy-in of results.	Interpretation of results is variable, modelling can be applied without consideration of biology and context. Some uncertainty in use of habitat suitability curves as predictors – particularly around invertebrates. Habitat analyses rely on good information on habitat suitability.
2D Hydraulic model and habitat evaluation	Predicts water depth, velocity, and habitat suitability as a function of flow.	When working outside boundaries of current wetted channel, extrapolating beyond calibration data provides good 2D graphics for visualisation of predictions. Can extrapolate further than 1D model, especially suitable for braided rivers and rivers where flow patterns change significantly with flow (eg, overbank flows).	Requires significant and expert data inputs and analysis, difficult and expensive to apply on shallow boulder rivers. Interpretation of results is variable, modelling can be applied without consideration of biology and context. Habitat analyses rely on good information on habitat suitability, and limitations of habitat model are not always understood. Some uncertainty in use of habitat suitability curves as predictors – particularly around invertebrates.

Rivers and streams	Description	Pros	Cons
Connectivity/fish passage assessment	Habitat method applied in a critical reach, identified by survey. See 1D and 2D hydraulic models above.	Addresses specific issue at specific downstream locations.	Need to survey entire river to identify critical reach for modelling, requires significant field work input, biological interpretation can be difficult, don't know what length of time is sufficient for fish passage nor how length of critical reach (eg, critical riffle) interacts with critical passage depth.
Entrainment model	Predicts water levels for high flows. Critical flows for moving bed sediments based on 1D or 2D hydraulic models (above).	Most relevant for rivers with major impoundments or large diversions where there is opportunity to manage flows, provides flushing flow requirements, stability of habitat, important base for ecosystem analysis.	Need accurate estimates of bed sediment size, internationally science is not very accurate and still evolving, not a precise science, needs more validation of accuracy of predictions. Quite specialised, requiring hydraulic expertise. Validation important but rarely carried out.
Sediment transport	See entrainment model		
Bank stability	See entrainment model		Specific assessment of bank stability or erosion complex and difficult.
Suspended sediment	Predicts how downstream sediment concentrations vary with flow. Particle settling model using 1D hydraulic model above.		May need suspended sediment measurements for calibration and verification.
Seston flux	Applied in special circumstances, dispersion settling model used in lake outlets to determine how flow change will affect seston distribution. See 1D hydraulic model above.	Specifically takes account of lake outlet circumstances, which can support highly valued fisheries, demonstrates value of water itself (ie, it carries in suspension plankton etc, which is utilised by filter feeding invertebrates, which in turn are food of fishes).	May need biological data for calibration and verification.

Rivers and streams	Description	Pros	Cons
Inundation model	Predicts area of inundation and time, in relation to flood flows. See entrainment model above.	Identify critical bands of inundation flows/levels, allows protection of wetland habitat (interpretation is difficult), protects property.	Data requirements are extensive (surface roughness, survey), quite specialised, biological interpretation is difficult.
Flow variation analysis	Identification of critical features of flow variability to maintain ecosystem functions, often based on results of other models (eg, entrainment, fish passage) .	Allows for broad ecosystem requirements that may not be picked up in specific habitat analysis.	Science is still evolving, difficult to set flow thresholds except for some ecosystem functions (eg, river mouth opening for fish recruitment).
Fish bioenergetics model	Spatially explicit predictions of net rate of energy intake fish growth potential fish positions, and overall carrying capacity of a function of flow. Uses 1D representative reach or 2D hydraulic models (see above).	Provide biologically meaningful predictions that user groups can relate to (good 2D graphics for visualising predictions), outputs useful educationally, and quantitatively links hydraulics, invertebrate drift (fish food) and salmonid foraging processes.	Complicated and expensive. Available for only one species (brown trout), but could be parameterised for other salmonids.
Water quality models	Includes temperature and dissolved oxygen. Uses generalised habitat or 1D hydraulic model above.	Requires some data and links flow to critical parameters (temperature and dissolved oxygen). Application is relatively simple (eg, WAIORA).	Complicated to calibrate model, requires training in application of principles.
Groundwater models	Predicts effect of groundwater abstraction on surface water flow.	More holistic evaluation of water movement and effects.	Currently a research area, needs verification, specific to porous alluvial rivers, very detailed assessments required, data requirements are high/expensive.

2.5 Description of Individual Methods

In Section 2.3 we described the framework for method selection, which was based on the significance of instream values (Section 2.1.1) and degree of hydrological alteration (Section 2.2). In this section we describe all methods and make recommendations for their use within the context of the framework. A stylised representation of the framework (Table 2.4) is given with each method to assist the reader in placing the method within the context of the framework.

2.5.1 Historical Flow Methods

Framework for use (Table 2.4):

Hydrol. alteration	Values L	M	H
L	✓	✓	
M	✓		
H			

Historical flow 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 based on a proportion of a flow statistic to specify a minimum flow. The statistic could be the mean annual low flow, a percentile from the flow duration curve, or an annual minimum with a given exceedance probability (see Historical flow method 1 in Table 2.7). For example, a method might prescribe that abstraction ceases when the natural flow falls below 80% of the MALF. Another method that has been used is to allow the total amount of water taken from the river to vary with the flow, eg, allow abstraction of 10% of the flow at any time (see Historical flow method 2 in Table 2.7).

The aim of historical flow methods is to maintain the flow within the historical flow range, or to avoid the flow regime from deviating largely from the natural flow regime. The underlying assumption is that the ecosystem has adjusted to the flow regime and that a reduction in flow will cause reduction in the biological state (abundance, diversity, etc) proportional to the reduction in flow; or in other words, that the biological response is proportional to flow (Figure 2.3). 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.

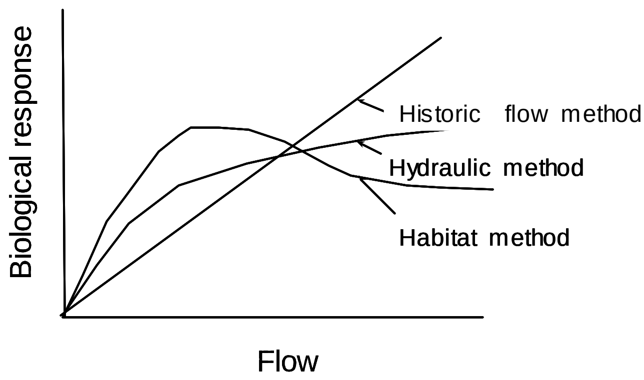

Figure 2.3: Hypothetical relationships between assumed biological response to flow for the historical flow, hydraulic and habitat methods. The biological response is assumed to be proportional to the flow, the wetted perimeter or width, and the weighted usable area – for the historical flow method, the hydraulic method, and the habitat method, respectively.

The most well known historical 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 streams (in Montana, Wyoming and Nebraska) and an assessment of the depths and velocities needed for sustaining aquatic life. At 10% of average flow, he found that the average depth was 0.3 m and velocity 0.25 m/s, and considered these lower limits for aquatic life. He found that 30% of average flow or higher provided average depths of 0.45–0.6 m and velocities of 0.45–0.6 m/s and considered these to be in the good to optimum range for aquatic organisms. This is an example of a 'regional method', applicable to the region that has the same type of streams as the streams used for developing the method. 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 (eg, Allan 1995; Crowe et al. 2004). In New Zealand, Fraser (1978) suggested that the Tennant method could be extended to incorporate seasonal variation by specifying monthly minimum flows as a percentage of monthly mean flows.

Historical flows can also be used to define 'an ecologically acceptable flow regime' – for example, Arthington et al.'s (1992) 'holistic method' that considers the magnitude of low flows, and the timing, duration and frequency of high flows. Such a flow regime would not only sustain biota during extreme droughts, but would also provide high flows and flow variability needed to maintain the diversity of the ecosystem. The building block method (BBM: King et al. 2000) is a similar approach. The range of variability approach (RVA) and the associated indicators of hydrological alteration (IHA) identify an appropriate range of variation, usually one standard deviation, in a set of 32 hydrological parameters derived from the 'natural' flow record (Richter et al. 1997). The holistic, BBM and RVA methods are conservative and maintain the ecosystem by retaining the key elements of the natural flow regime. They are probably most appropriate for river systems where the linkages between ecosystem integrity and flow requirements are poorly understood.

Recommendation: Historical flow methods can be used when the degree of hydrological alteration is low and when values are low to medium, or when values are low and the degree of hydrological alteration is low to medium.

2.5.2 Expert Panel

Framework for use (Table 2.4):

Hydrol. alteration	Values L	M	H
L	✓	✓	
M	✓		
H			

Expert panels have been used by some regional councils in New Zealand. These usually comprise interested parties as well as 'experts'. This method has been used to support or verify other methods of ecological flow assessment, rather than as a method in its own right. The expert panels inspect the stream from the banks and consider the suitability of suggested ecological flow requirements. If the stream is at or close to the suggested ecological flow, it is possible to assess its suitability for many aquatic biota, provided the panel has the relevant experience. However, if the stream is not close to the ecological flow it is very difficult, if not impossible, to envisage hydraulic conditions at other flows.

Recommendation: Expert panels are most useful for assisting in gaining consensus where there has already been an ecological flow assessment made by another method. Its effectiveness is limited by the credibility of the experts, it is not quantitative or objective, and the need for consensus can lead to inaccurate outcomes. It is recommended as a method in its own right only when instream values and degree of hydrological alteration is low to medium.

2.5.3 Generalised Habitat Models

Framework for use (Table 2.4):

Hydrol. alteration	Values L	M	H
L			✓
M	✓	✓	
H	✓		

Studies of flow and habitat requirements in more than 60 New Zealand rivers suggest that flow requirements can be generalised for particular species (Jowett 1996). For trout, these generalised relationships vary with fish size and life stage. Trout rivers, even of the same size, vary in the value of the fisheries they support. Moreover, different sizes of trout and life stages have different depth and velocity and related flow requirements. Jowett's studies indicate that maximum habitat for juvenile trout tends to be provided by flows of 1–2 m^3/s, whereas maximum habitat for adult brown trout is provided by flows of 6–15 m^3/s. More recently, a larger set of rivers (99) was examined by Lamouroux and Jowett (2005) to show that the shape of the relationships between habitat and flow per unit width was consistent between rivers. The results of this analysis allow us to more closely define the flows that provide maximum habitat for each species, and more importantly, quantify the habitat change that occurs with flow change.

The habitat suitability curves used to generate the generalised habitat models were regarded as the best currently available (2004), but further refinement is possible as more data on habitat use are collected or more refined methods of analysis are developed.

Generalised models can be applied to a specific stream with simple spreadsheet calculations while knowing only the average width at one discharge. This assumes that the hydraulic geometry (relationship between width and flow) is typical of New Zealand rivers, as described in Jowett (1998). However, there are limitations. The generalised models were based on 99 New Zealand streams and rivers and so represent the results of habitat analysis in a river of 'average' shape. This assumption breaks down where a river is unusually wide and shallow, as in extensively braided rivers, or where the river is narrow and deep, as in some spring-fed streams. If flow and width measurements are carried out at two flows, WAIORA (Water Allocation Impacts on River Attributes) can be used to apply the generalised relationships with greater certainty. The application and data requirements are described in the WAIORA user guide (Jowett et al. 2003).

Recommendation: Generalised habitat models are a step up in precision from historical flow methods and so should be used when the degree of hyrological alteration or values are higher. These models have advanced sufficiently to be applied nationally – but testing and refinement at the regional level is recommended.

2.5.4 Fish Passage/Connectivity Models

Framework for use (Table 2.4):

Hydrol. alteration	Values L	M	H
L			✓
M		✓	✓
H	✓	✓	✓

Fish passage can be an issue at low flows when there is inadequate water depth for upstream/downstream migration of fish species or laterally into wetlands and side braids. Fish migration is often related to spawning and is a seasonal requirement. Survey reaches and cross-section locations are selected that represent potential barriers to fish passage. Hydraulic modelling is carried out to determine the lowest flow that provides adequate passage width through the river. Passage width is the continuous width of river with depth exceeding the minimum passage depth and velocity less than the maximum passage velocity. These models are usually applied to salmonids because these fish have greater depth passage requirements than native fishes.

Recommendation: Fish passage/connectivity modelling should be undertaken whenever passage of valued fish species is a significant issue identified by stakeholders. It would usually complement routine 1D and 2D hydraulic/habitat modelling.

2.5.5 1D and 2D Hydraulic Habitat Models

1D models

Framework for use (Table 2.4):

Hydrol. alteration	Values L	Values M	Values H
L			✓
M		✓	✓
H	✓	✓	✓

2D models

Framework for use (Table 2.4):

Hydrol. alteration	Values L	Values M	Values H
L			
M			✓
H		✓	✓

In contrast to standard setting methods, hydraulic-habitat methods predict how the river changes incrementally with flow and hence are most suited to evaluating the effects of large-scale changes to the natural flow regime. As such they enable a holistic approach to be taken to the assessment of flow regime requirements, where necessary. Hydraulic-habitat methods are a primary component of the IFIM. Data requirements and hydraulic modelling capability increases with the complexity of the underlying hydraulic models. However, the biological interpretation of results is critical and the process allows other aspects of flow effects on biota (eg, duration of low flows, flow variability and frequency of flushing flows, food supply, and passage restrictions) to be considered, as far as possible with existing knowledge.

At the least, the basic premise of these methods is to maintain instream habitat that is suitable for the biota. The critical values and their associated habitat suitability curves must be appropriate to the stream, particularly its size, and must be related to flow, particularly minimum flows, if hydraulic-habitat models are to produce consistent and sensible results; and to provide a context for connected systems such as riverine wetlands or groundwater systems.

The critical factors and their associated habitat suitability criteria can be perceived in two ways. In most instances, we apply them in a specific sense for providing habitat for the target critical species/life stage and with the added aim of providing for taxa with lower flow requirements. However in some situations, habitat criteria associated with the critical factor can be used in a generic sense to provide instream conditions that, based on experience, are considered appropriate for the ecological function and potential range of instream communities. In this latter situation, the habitat criteria act as general descriptors of instream conditions and stream size; the 'target species' is secondary and may in fact not actually be present. Examples of these applications include:

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

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

Hydraulic modelling is used to predict water depths and velocities at individual points in a section of river over a range of flows. These predictions are then used to show how usable habitat varies with flow. Ecological flow assessments are based on the shape of the curves and the proportional changes engendered by a flow change. Either 1D or 2D modelling can be used, and if done well, there should be little difference between the results. 2D modelling can only be applied to a reach, the length of which is usually up to 1 km, a constraint imposed by survey costs. The reach is usually chosen to represent a longer segment of river.

The difficulties in acquisition of sufficient and accurate bed topography and calibration of 2D models are a practical limitation to their utility, and it should not be assumed that they are better simply because they require more data. A good knowledge of hydraulics is necessary to identify salient features of bed topography, especially in turbid or deep water. Calibration is difficult, subjective, and time-consuming with large files. In contrast, 1D survey methods are straight-forward and calibration procedures are well-developed and reproducible, although empirical. 1D surveys can be carried out over longer sections of river using the habitat mapping method, so that they can include a greater variety of habitats, although not at the same level of detail as a 2D survey. 1D surveys require fewer resources than 2D surveys, and usually produce similar or better accuracies. However, 2D models are better able to extrapolate beyond the calibration range in complex river morphologies, give good graphic representation, and when the modelling is done well, they give better predictions of the direction and distribution of velocities.

It is difficult to calibrate a 2D 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. This becomes particularly critical at low flows, where the definition of upstream and downstream water level controls, such as topography at the head of a braid, or topography of a riffle at the tail of a pool, determines the flow in the braid or the water level in pool. Thus, the accuracy of the topographic model will determine whether water levels are predicted correctly at low flows.

1D models using empirically derived stage-discharge rating curves 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. 2D models are likely to predict changes in velocity distribution, particularly eddy and reverse flows, more accurately than 1D models, although in both cases, predicted depths and velocities will be incorrect if water surface levels are not modelled accurately.

The number of cross-sections in a 1D survey depends on the morphological variability within the river. An analysis of a 2D survey showed that similar results could be achieved with a 1D survey with 19 cross-sections (Tarbet and Hardy 1996). In another study, Payne et al. (2004) sub-sampled several very large data sets to determine how many cross-sections are required to produce a robust weighted usable area function. They found that 18–20 cross-sections gave results nearly identical to results for 40 to 70 cross-sections per reach.

As with all methods listed, instream habitat surveys and analyses have to be carried out appropriately. The main factors in habitat analyses are that:

1. the physical habitat measured is representative
2. the aquatic organisms may or may not be habitat-limited, but it is conservative to assume that a population is habitat-limited. The preference (habitat suitability) curves should reflect the preferred habitat of the organisms
3. suitability curves should be based on measurements that show how distributions of abundance or presence of organisms vary with physical habitat. If measured abundance or distribution does not vary with physical habitat, habitat methods are not applicable.

Further discussion on hydraulic habitat models is found in Appendix 3.

Recommendation: 1D models are the most appropriate tools when the degree of hydrological alteration or values is high, or when both hydrological alteration and values are medium. Because of their greater expense and difficulty, 2D models ought to be confined to applications where channel geomorphologies are complex (eg, braided rivers), or when difficult surveying conditions require that model predictions are extrapolated substantially outside model calibration flows, or when spatially explicit predictions of hydraulic and habitat features would aid understanding by stakeholders and provide input to other models (eg, bioenergetics fish models). They are appropriate when the degree of alteration or values is high to medium or when both hydrological alteration and values are high.

a. Regional methods

Framework for use (Table 2.4):

Hydrol. alteration	Values L	M	H
L			✓
M	✓	✓	
H	✓		

Regional methods are a *subset* of hydraulic habitat methods and are based on an analysis of ecological flow assessments made by habitat-based methods in a number of rivers. They have already been developed for some areas of New Zealand (eg, Jowett 1993a, b; Wilding 2002). Typically, the ecological flow requirement will be a function of river size. Regional methods are quick and easy to use, once they have been developed, and are almost as biologically defensible as the assessments on which they are based.

Tennant's (1976) method is a good example of a regional method that combines the best features of historical flow methods and habitat methods, resulting in a biologically defensible method of minimum flow assessment – for the region. Once established, regional methods can be easily applied to rivers within the region using a formula based on the proportion of natural flow, either recorded or estimated. The formula can be as simple as a fixed proportion of flow or can vary the proportion with river size, possibly retaining a higher proportion of the flow in small rivers than in larger rivers, as used in formulae for maintenance of trout and food-producing habitat in Wellington and Taranaki rivers (Jowett 1993a, b). Similar methods could be developed for regions that are hydrologically and morphologically similar, with criteria that apply to trout, native fish, stream insects, or periphyton.

By analysing habitat variation with flow for rivers within a region, it is possible to determine the level of flow as a proportion of median or mean annual low flow that maintains adequate or optimum conditions for various 'target' communities. Variation in levels of maintenance could be achieved by assessing requirements for optimum habitat and minimum habitat, as in the Tennant method. Application of the method would involve selecting an appropriate target community and level of maintenance for the river in question and then applying a formula based on flow. The formula may be referenced to an historical flow statistic (eg, MALF: Jowett 1993a,b).

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

The rationale for habitat-based regional methods is primarily that of habitat methods. Within a region, it is possible to develop formulae 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. They are less expensive than habitat methods, yet once developed are cheaper but still likely to result in flow assessments that provide life sustaining flows whilst retaining some degree of the river's 'character'.

Recommendation: Regional methods would be used in similar circumstances as generalised models. Like generalised habitat models they offer a low to modest cost option for setting minimum flows, with greater precision than historical flows, when the degree of hydrological alteration or values are higher. However, generalised habitat models have advanced sufficiently to supersede regional methods.

2.5.6 Periphyton Biomass Model

Framework for use (Table 2.4):

Hydrol. alteration	Values L	Values M	Values H
L			
M			✓
H		✓	✓

This is a specialised model that would mainly be applied to ecological flow assessments where the frequency of floods and freshes is altered. The model predicts accumulated periphyton biomass from the time since the last flood and nitrogen and phosphorus concentrations (as mean monthly concentrations measured over at least a year). The model is based on an analysis of periphyton samples collected in a large number of rivers and is described in Biggs (2000). In rivers below impoundments, this relationship can be used to develop the necessary frequency of artificial floods for various growth periods to ensure that peak biomass does not exceed biomass guidelines.

Recommendation: Periphyton biomass models should only be used as part of an ecological flow assessment, and only to predict frequency of artificial freshes below an impoundment to manage periphyton growths below the target maximum abundance.

2.5.7 Entrainment, sediment and bank stability models

Framework for use (Table 2.4):

Hydrol. alteration	Values L	M	H
L			
M			
H		✓	✓

The models are used to assess flushing flow requirements, where flows should move fine sediments, but not the armour layer and channel maintenance flows, where a significant portion of the armour layer is disturbed. Entrainment models are an extension of hydraulic models and would only be applied to ecological flow assessments where the frequency of floods and freshes is altered. These models are used to predict bed shear stress and velocities and hence potential movement of bed sediments at high flows. The output of these models describes how the area of river with shear stresses exceeding critical shear stresses for bed sediment movement varies with flow. For bank stability, shear stresses at the banks are compared to critical shear stresses. Critical shear stresses for non-cohesive sediments are relatively well known (eg, Graf 1971). Critical shear stresses for cohesive sediments are reviewed by Jowett and Elliott (2006).

Recommendation: Entrainment models are an extension of hydraulic models and would only be applied to ecological flow assessments where the frequency of floods and freshes is altered – such as occurs with large abstractions, diversions, or impoundments (ie, high degree of hydrological alteration). These models are used to predict bed shear stress and velocities and hence potential movement of bed sediments at high flows.

2.5.8 Suspended Sediment Model

Framework for use (Table 2.4):

Hydrol. alteration	Values L	M	H
L			
M			✓
H		✓	✓

This model, sometimes known as the 'sticky bed model' predicts how suspended sediment concentration varies with distance downstream under different flows (Jowett and Milhous 2002). The model can be used to determine downstream changes in suspended sediment concentration (water clarity) that result from water abstraction or diversion. The assumption is that the stream bed is a matrix of gravel and cobbles and that any suspended sediment reaching the water/substrate interface will be trapped within the matrix. This mechanism has been shown to be valid until all voids in the substrate matrix are filled or a surface seal forms. Calibration measurements of flow and suspended sediment concentrations downstream of the abstraction point are advisable but not essential. Currently, river hydraulic habitat simulation (RHYHABSIM) is the only hydraulic model that does this calculation. This type of model would only be applied to relatively large abstractions or diversions; applications and field measurements of suspended sediment concentration indicate that the changes in sediment concentration are small and barely detectable.

Recommendation: The suspended sediment model is a specialised model that would only have application to large abstractions or diversions (high degree of hydrological alteration). It would form part of the ecological flow assessment where there is reason to believe that the hydrological alteration would significantly change the suspended sediment regime.

2.5.9 Seston Flux Model

Framework for use (Table 2.4):

Hydrol. alteration	Values L	M	H
L			
M			✓
H		✓	✓

The seston flux model is the suspended sediment model described above, applied to the special case of seston (plankton) flowing from a lake. Lake outlets commonly support high densities of filter-feeding benthic invertebrates and seston is their main food source. Flow affects the distance that seston is carried downstream and effective length of 'lake outlet'. As with the suspended sediment model, field measurements of seston concentration at a number of points below the lake outlet should be taken to calibrate the model.

Recommendation: The seston flux model is a specialised model that would only have application to large abstractions or diversions, where seston is likely to be impacted by the amount of large abstractions, diversions, or impoundments (high degree of hydrological alteration), and by the distance (from a lake outlet) it was transported down a river.

2.5.10 Inundation Models

Framework for use (Table 2.4):

Hydrol. alteration	Values L	M	H
L			
M			
H		✓	✓

Inundation models are hydraulic models, usually 2D, that show the extent of inundation of wetlands or similar riparian zones for different flows. The model would only be applied to ecological flow assessments where the frequency of floods and freshes is altered. The model can be used to determine critical flows for wetland maintenance or could be used for flood hazard mapping for damage avoidance.

Recommendation: Inundation modelling is a specialised application that would only be used as part of an ecological flow assessment where inundation of riparian and wetland area is likely to be significantly altered as a consequence of a change in the frequency of floods and freshes.

2.5.11 Fish Bioenergetics Models

Framework for use (Table 2.4):

Hydrol. alteration	Values L	M	H
L			
M			✓
H		✓	✓

These models have been developed for brown trout in New Zealand and the United Kingdom and for rainbow trout in the United States, but have not been widely applied. They are an extension of habitat models in that they assess the habitat suitability for drift-feeding fish taking into consideration factors such as drift-food availability, swimming ability, foraging behaviour, and metabolic processes. Both bioenergetic models and drift-feeding habitat suitability models predict trout feeding locations, with the bioenergetic model providing an alternative to the empirical habitat suitability observations. Comparison of New Zealand brown trout bioenergetic and habitat suitability models shows excellent agreement.

The advantage of bioenergetic models is that they predict biological meaningful metrics such as net rate of energy intake, growth potential, and carrying capacity for trout, in graphical outputs that are easily understood by stakeholders. This is a significant advance over weighted useable area (WUA) – flow relationships predicted by traditional 1D or 2D habitat modelling based on empirical habitat suitability criteria. However, at present they can be applied only at limited spatial scale (eg, over a riffle, pool sequence), and are expensive. The bioenergetic model developed at the Cawthron Institute operates on the output of 2D or 1D representative reach hydraulic models (Hayes et al. 2003; Kelly et al. 2005). Another similar bioenergetics based model developed in the United Kingdom operates on the output of a 3D hydraulic model (eg, Booker et al. 2004).

Recommendation: Fish bioenergetics models are new tools emerging from research and development. Although they require more testing, even at this stage they offer useful biologically based insights into the effects of flow change on salmonids. They are appropriate as a complement to 1D and 2D hydraulic/habitat models in situations where the degree of hydrological alteration and salmonid fishery value is significant.

2.5.12 Dissolved Oxygen Models

Framework for use (Table 2.4):

Hydrol. alteration	Values L	M	H
L			
M			✓
H		✓	✓

Dissolved oxygen concentrations can fall below acceptable levels in low-gradient streams that contain macrophytes or decomposing organic matter on the bed; in these slow-flowing streams the flow required to maintain an adequate dissolved oxygen concentration is an important ecological consideration. Three important parameters, as well as habitat and water temperature, are required to calculate flow effects on dissolved oxygen concentration. These are:

- daily community respiration rate (the average rate of oxygen consumption by aquatic plants and micro-organisms)
- production/respiration ratio (ratio of the daily rates of photosynthetic production of oxygen to daily oxygen respiration by plants and micro-organisms)
- reaeration coefficient (the coefficient that describes the rate at which oxygen is exchanged between the atmosphere and the stream).

The WAIORA DO model applies to streams with a reasonably homogenous distribution of aquatic plants (which can include algae) in a reach. At present, we do not have a model that applies to streams where low concentrations of dissolved oxygen are caused by anaerobic decomposition of organic matter. Model calibration using field measurements of dissolved oxygen, stream flow, width, depth, water temperature, and climatic conditions is essential, because modelling parameters are difficult to estimate.

Recommendation: Dissolved oxygen models such as WAIORA should only be used as part of an ecological flow assessment in low-gradient rivers, where changing the flow regime is likely to lead to a significant change in the density of aquatic macrophytes or filamentous algae, which in turn could result in increased frequency of diurnal dissolved oxygen depletion due to respiratory activity.

2.5.13 Temperature Models

Framework for use (Table 2.4):

Hydrol. alteration	Values L	M	H
L			
M			✓
H		✓	✓

Water temperatures may affect aquatic systems in many ways ranging from acute lethal effects to chronic stresses. When the flow of a river is reduced, it becomes more responsive to solar radiation because it is shallower and flowing more slowly. Water temperature modelling is soundly based on physical principles and a calibrated water temperature model is capable of predicting flow effects accurately. The water temperature models used to assess flow effects are usually one-dimensional heat transport models that predict water temperatures from the abstraction point as a function of stream distance downstream and environmental heat flux. In general terms, the heat is gained or lost from a parcel of water as it passes through a stream segment. This is accomplished by simulating the various heat flux processes that determine that temperature change. These physical processes include convection, conduction, evaporation, as well as heat to or from the air (long-wave radiation), direct solar radiation (short-wave), and radiation back from the water.

The temperature of water in a river is influenced more by climate than by river flow. The effect of water temperatures on aquatic biota is difficult to determine, largely because lethal water temperatures only occur for short periods when climatic conditions are extreme and flows low. Because of the dependence of water temperature on climate, it is difficult to use water temperatures to set ecological flow requirements, although it is possible to quantify the temperature changes that occur.

Recommendation: Temperature models should be used as part of ecological flow assessments for large-scale alterations which may result in heating of water in an affected reach, above that tolerated by fish and invertebrates. An example might be below a water supply impoundment.

2.5.14 Groundwater Models

Framework for use (Table 2.4):

Hydrol. alteration	Values L	M	H
L			
M			✓
H		✓	✓

Groundwater abstractions can influence the surface flows in a river, depending on the hydro-geology and the distance of the groundwater take from the river. Similarly, changes in river water levels can influence groundwater levels. Groundwater models are discussed in the Groundwater section of this report.

Recommendation: Groundwater models should form part of the ecological flow assessment of surface waters in situations where groundwater abstraction has the potential to significantly alter the flow regime of a river and where the degree of hydrological alteration and/or values are high.

3 Lakes and Wetlands

3.1 Ecological Values, Factors and Principles

3.1.1 Values

a. Lakes

In lakes, the hydrological regimes that influence inflows and outflows result in changes to lake levels, and levels rather than flows per se have the greatest influence on lake ecosystem values, especially in larger lakes. Water level fluctuations control aquatic biota in lakes primarily through their effects on the species composition, distribution and productivity of the shallow-water littoral zone communities. In addition, the inflow-outflow regime influences the lake water residence time, a key parameter in setting lake water quality over long time frames. Changes in residence time are most likely to affect smaller lakes. A reduction in through-flow and increase in residence time increases the likelihood of algal blooms. At the other extreme, large decreases in residence time can reduce the productivity of lake communities.

Rules for hydrological management for lakes should be based on permitted ranges in water level and rates of water level fluctuation that protect lake values. There have already been cases where local communities have identified lake values to be protected by management rules. In the case of Lake Taupo (Taupo-nui-a-Tia 2004), 12 values were clearly identified and recognised, especially clear water, diverse plants and animals, good trout fishing, high-quality inflows and a weed-free lake. The guidelines for the protection of Lake Manapouri relate particularly to water level fluctuations in the context of hydro-electric extraction, and these can be applied to other lakes where water abstractions may affect lake level and shoreline fluctuations. Table 3.1 lists ecological values associated with lakes that are related to water levels.

Table 3.1: List of lake values and factors related to water levels

Values/management objectives	Possible factors that affect ability to achieve objective
Salmonids	Connectivity of migration pathways to spawning streams Rearing habitat in littoral zone Habitat of food sources Extent of littoral zone for adult habitat Lake stratification as it affects dissolved oxygen and temperature Water quality (clarity and sediment concentration)
Native fish	Spawning habitat Rearing habitat in littoral zone Habitat of food sources Extent of littoral zone for adult habitat Connection / frequency of flow to riparian area, wetlands and spawning streams Lake stratification as it affects dissolved oxygen and temperature Water quality (clarity and sediment concentration)
Benthic invertebrates	Spawning habitat Rearing habitat in littoral zone Habitat of food sources Extent of littoral zone for adult habitat Substrate Riparian vegetation Water quality (dissolved oxygen and temperature)
Submerged macrophytes and algae	Littoral zone substrate Littoral zone depth profile Water clarity
Emergent plants	Littoral zone substrate Littoral zone depth profile
Planktonic communities	Lake stratification as it affects dissolved oxygen, temperature and nutrients Water quality (clarity and sediment concentration)
Birds	Littoral zone feeding and sheltering habitat (see emergent plants/macrophytes above)
Water quality (intrinsic value to humans)	Clarity
Riparian	Lake edge erosion, slumping and wave action Soil moisture / water availability for riparian plants
Foreshore	Drawdown zone Frequency of water level fluctuations

b. Wetlands

Hydrology is the fundamental driver of all ecological processes in wetlands and directly or indirectly controls all aspects of nutrient cycling and availability, water level, primary and secondary productivity, and habitat availability (eg, relative areas of vegetated habitat versus open water). Water level fluctuations are particularly critical for ecological values and the controls on the distribution of organisms in most wetlands. Aspects such as the depth and duration of flooding periods, depth to water table, and duration of drawdown periods when the water level is below the soil surface, are those that control distributions of organisms in most wetlands. Flow velocity may directly affect some wetlands that are very closely linked to a river channel, but most riverine wetlands are relatively quiescent backwaters or adjacent depressions that are indirectly controlled by groundwater inputs and overland flooding.

Vegetation is sensitive to water regime because plant species differ in their tolerance to flooding. Species diversity is generally highest in wetlands with moderate water level fluctuations. It decreases if the water level remains constant or fluctuates widely. For individual wetlands, a variety of water depths within a site, with a mixture of open water and shallows, allows emergent and submerged plants to grow. The water regime is also very important for preventing terrestrial weed invasion and limiting mammalian predator access, giving the wetland system its robustness to withstand external pressures. Many examples of loss of natural character in New Zealand wetlands relate to water abstraction and drainage that lower water tables, allowing competitive terrestrial flora to invade and displace flood-tolerant wetland species. The effects of water levels on the structure (ie density, height, and physical complexity) of lake littoral zones and wetland vegetation are important for invertebrate and periphyton values, and affect the availability of habitat for fish and birds. Different bird species, in particular, have different vegetation preferences for nesting and feeding.

There are other ways in which the water regime can control biotic composition. The timing and duration of the connectivity of wetlands with their parent lake and rivers control migration patterns of fish and breeding cycles of birds. Wetting and drying cycles determine the hatching of invertebrates, the flowering of many plants and the availability of areas of shallow open water for wading birds.

Table 3.2 lists values associated with wetlands and related to water levels.

Table 3.2: List of wetland values and factors related to water levels.

Values/management objectives	Possible factors that should be considered
Fish (native and salmonids)	Connectivity between wetland and main channels Habitat of food sources Submerged vegetation for habitat Wetting and drying cycles Water quality and temperature
Benthic invertebrates	Wetting and drying cycles for hatching Sediment transport Periphyton and detritus (ie, food) Vegetation structure Substrate Water quality (pH, dissolved oxygen and temperature)
Submerged macrophytes and algae	Substrates (including plants for epiphytic algae) Topography (availability of deeper pools) Water clarity pH and nutrients
Emergent plants	Water level fluctuations Predominant water source (rainwater, groundwater, surface flow) Substrate Nutrients
Birds	Wetted area Vegetation structure Habitat for food sources Predator access

3.1.2 Principles for Determining Significance of the Values

a. Lakes

The significance of lake values determines the level of protection lakes are given and the methods used to assess flow and level requirements for maintaining their values.

A classification of New Zealand lakes similar to the River Environment Classification (currently under development) may form a further basis for setting values. This work is ongoing, but a recent snapshot of New Zealand lake water quality' (Sorrell et al. 2006) produced for the Ministry for the Environment provides a first step in comparing the state of a lake's quality in relation to regional and national averages.

When considering ecosystem values of lakes, there are two distinct zones that respond quite differently to changes in lake levels. These are the littoral zone and the pelagic – open water or pelagial – zone (Figure 3.1). The littoral zone is the shallow-water zone around the edges of lakes (in contrast to the open water pelagic zone) that is occupied by submerged plants attached to the bottom. The littoral zone is characterised by high biodiversity and multiple ecosystem functions and is the important interface between land and lake. It can easily be envisaged from Figure 3.1 that the littoral zone will be affected by lake level variation. The situation is exacerbated in turbid lakes where the depth of the littoral zone is restricted by low light penetration.

The shape of the lake basin, the lake shoreline length and the water clarity dictate the extent of the littoral zone versus the pelagic zone. In deep steep-sided lakes, the littoral zone may be relatively less important in overall lake functioning than in shallow lakes with gently sloping shorelines. Even in the former, there may be distinct and specific high values associated with individual species in littoral zones (eg, spawning sites for some fish species, cover and nesting sites for bird species).

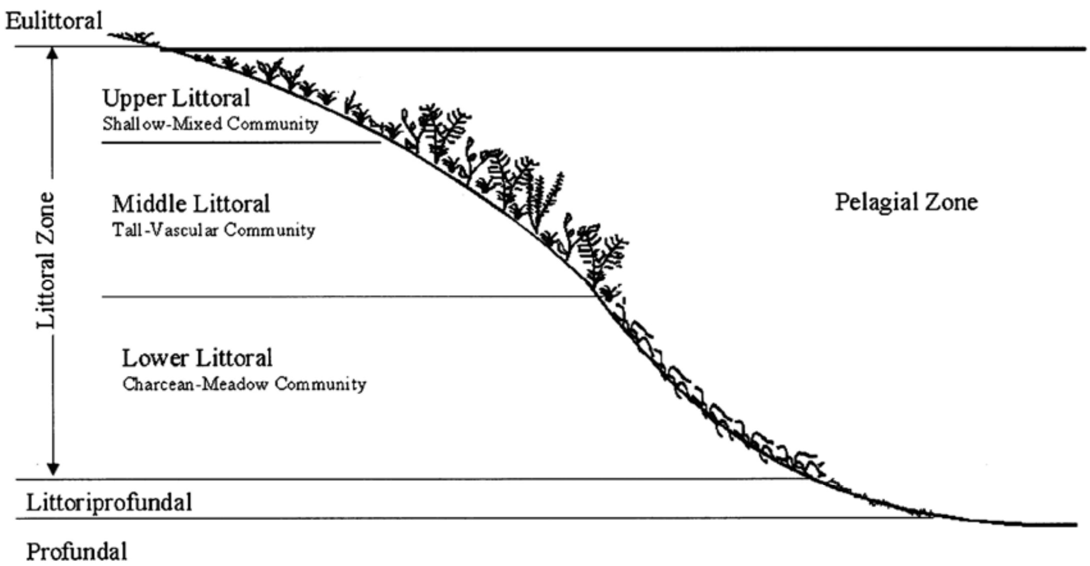

Figure 3.1: Diagram of the littoral zone in most deep (>40 m) New Zealand lakes, showing the various plant communities (from: Kelly and McDowall 2004).

Wave-cut platforms are common features in large lakes where wave action is significant and the bottom profile of the littoral zone is modified by waves. These are commonly ideal areas for the development of a diverse and productive littoral community and their characteristics (and existence) depends on the interactions of waves and lake levels. Any alteration to lake levels in such situations may have profound effects on lake edge ecosystems. It should be recognised that in many cases when lake levels change over long time periods (ie, a long-term change in mean lake level) littoral communities may adapt over time to the new levels by moving up or down the lake bed profile. Alternatively, if the magnitude in the fluctuation of lake-levels changes at shorter (seasonal) time scales, this can have adverse effects on littoral zone communities (James and Graynoth 2002).

Natural lakes with high water quality are often those with relatively high oxygen concentrations in the lake water column throughout the year. Maintenance of oxygen throughout the water column ensures habitats for a wide variety of organisms. Oxygen is maintained through wave action and wind mixing, and through inflows that sink to the lake bed while carrying dissolved oxygen down with them.

b. Wetlands

Different types of wetlands have characteristically different hydrology. Rainwater-dominated wetlands such as peat bogs rarely have standing water, and the water level is usually close to the surface, responding primarily to rainfall events. Groundwater-fed wetlands will respond to adjacent rivers, but often with pronounced time lags between changes in river flow and changes in wetland water level. Wetlands with surface connections to rivers may mimic the river fluctuations closely, or there may be sudden changes in wetland water level, eg, if the river needs to overtop natural or artificial banks and levees before the wetland floods, as applies to many floodplain wetlands.

Wetland ecological significance can be judged on factors such as national or regional significance, rarity, and representativeness. Wetland significance is judged against a background of widespread loss of wetland environments in New Zealand: over 90% of the pre-European wetland area has been lost to landscape development. Much of the remaining wetland area is alpine, and a disproportionate number of remaining sites are peaty bogs and fens not associated with rivers. Riverine wetlands are highly likely to have considerable significance in regions where large areas of wetlands have been lost. Wetlands are also important for values including hydrology, nutrient retention, recreation and culture. The historic loss and reduced current extent of wetlands in New Zealand justifies a very strict standard for any flow alteration that would lead to loss of wetland area or condition.

A number of resources are available for assessing significance of wetlands. An important first step in determining significance of a wetland is classification of the type of wetland (described by the MfE-sponsored classification scheme of Johnson and Gerbeaux (2004)). The condition of a wetland, which relates to its significance, can be assessed from the condition assessment methods of Clarkson et al. (2003). Important wetlands of New Zealand were compiled in the WERI (Wetlands of ecological and representative importance) database held by the Department of Conservation, recently updated as part of the Waters Of National Importance (WONI) project, which has also documented the historical extent of wetlands. The NIWA Freshwater Fish Database includes records from many wetland habitats. Botanical values of wetlands have been documented in many regional flora and there are many reports on specific wetlands produced for a range of local authorities. Landcare Research and NIWA are currently developing an environmental database that will allow species diversity of all major taxonomic groups to be related to wetland types, nutrient regimes and hydrological regimes. For assessment principles of representativeness, rarity etc, the methods of Whaley et al. (1995) were developed specifically for wetlands and have proven to be robust in a number of regulatory and environment court hearings.

As so much of the biodiversity value hinges on vegetation type and structure, maintaining vegetation properties is the most critical factor when assessing flow (ie, flooding and water level) requirements for wetlands. Wetlands can be very difficult to restore once soil drying has changed vegetation character, especially if large ‘transformer’ weeds such as willows have invaded. Connectivity requirements for fish access and habitat requirements for birds are the other main critical flow-related factors.

3.1.3 Principles for Determining Potential Change to Flow Regime/Levels

Lakes and wetlands are characterised by natural water level fluctuations. No natural lake or wetland has a precisely constant water level. Seasonal drawdown and recovery of water levels are ubiquitous, and there can also be wide inter-annual variability in water regimes. In the case of wetlands, at one extreme, there are ephemeral wetlands that remain dry for months or years at a time, with no apparent aquatic character, and flood irregularly. Even in permanent wetlands, water levels can disappear below the surface for prolonged periods, and standing water may only occur during brief floods. Many wetlands are a mosaic of depths, with permanent water in some areas, intermittently wet areas, and semi-wet margins that often support highly diverse mixtures of wetland and terrestrial species.

Variability in water level is what promotes the high species diversity in wetlands, and the littoral zones of lakes. This is demonstrated in Figure 3.2.

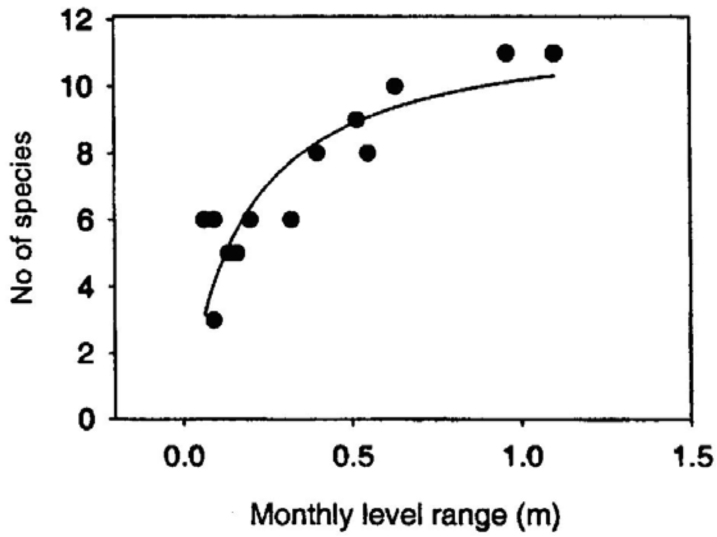

Figure 3.2: Relationship between water level range (mean monthly range) and the number of species of low-growing plants per lake (Riis and Hawes 2002: figure 2).

The key principle in predicting change is understanding the current water level variability of a site, how it relates to the current distribution of plants and animals, and how much the water level variability can change before species are lost or weeds and pests invade. The seasonal pattern of water levels (ie, the depth, timing and duration of standing water and drawdown below the surface) is termed the hydroperiod. The maintenance of a period of levels high enough to ensure connectivity between wetlands and adjacent water bodies is often fundamental to their existence as discrete ecosystems, and essential for migrations and other movements of fish that use both habitats.

Some knowledge of the existing hydroperiod is therefore essential in predicting the likely effects of flow abstractions or, in some instances, additions. Where available, historical records can be used very robustly to do this. Regular water level monitoring is a feature only of those lakes that are used for commercial or drinking water purposes. Water level monitoring is rare in New Zealand wetlands. Often it may be necessary to interpret the water regime from topographic information, soil types, and mapping of connectivity to rivers. This approach can provide a simple indication of water-biota interactions, but in any situation where the site is of high value or where large changes are likely, direct monitoring of hydrological characteristics before the proposed changes occur are essential. Methods for monitoring wetland water levels include capacitance probes and piezometers, weirs, flow sensors, dipwells and tipping gauges. Campbell and Jackson (2004) provide a useful overview of these techniques and their data interpretation. Multiple recorders are essential in order to characterise the water regime of different communities in larger wetlands with many vegetation types.

How much change to the hydroperiod can most wetland and littoral communities tolerate? Models developed for lakeshore turf communities and littoral macrophytes are likely to be applicable to wetland environments with submerged plants and algae. For the taller vascular plants, wetland communities are complex and how individual species change differs, depending on the existing water regime. In general for wetlands with predominantly subsurface hydrology, permanent changes of < 20 cm in water table depth are unlikely to have much effect on species composition, whereas lowering the water table > 30 cm from its existing level typically leads to changes in species composition. This is because the live roots of most wetland species occur mostly in the top 30 cm, and drying this zone allows terrestrial species to invade. In some sites the maximum depth over the season is the important feature determining species composition, in others it may be the median depth, or some other feature. Continuous water level records provided by equipment such as capacitance probes can be invaluable for making these assessments.

Many shallow-water plants that inhabit the littoral zone of New Zealand lakes require a dry period, when they are exposed to the air at some stage to allow for flower and seed production. In some locations this dry period may be only every few years, but in general, the shallow littoral zone of New Zealand lakes is adapted to some degree of water level variability. However, where the range in lake levels exceeds 2 metres and /or when the water levels vary too frequently, the littoral vegetation may disappear. For example, Lake Hawea has high clarity and an extensive littoral zone should be supported; yet it has a very limited littoral zone due to artificial water level fluctuations of greater than 8 m.

3.2 Determining the Degree of Hydrological Alteration

3.2.1 Lakes

The distribution and occurrence of healthy lake littoral habitats and communities varies with lake size, depth and water clarity. The risk that changing lake levels decrease available habitat or adversely affect communities depends on the lake bed profile (bathymetry), substrate type, water clarity, wave action as well as size and depth. The risks of deleterious effects are greater in shallower systems than in deep water bodies. Within a lake level range, impacts arise from changing seasonality in levels and the proportion of time spent at different levels (level duration).

Three parameters relating to lake level change are considered in Table 3.3. These are: median lake level (m), mean annual lake level fluctuation (difference (m) between maximum and minimum lake level), and seasonality of levels (seasonal pattern of relative summer vs. winter lake levels). Two types of lakes are distinguished: deep lakes (> 10 m maximum depth) and shallow lakes ≤ 10 m maximum depth) as the sensitivity to level alteration depends on the lake depth.

Table 3.3: *Descriptors of hydrological change for lakes.*

Risks under a potential change to flow regime may be defined for deep and shallow lakes as follows:	
Low risk	**Deep lakes (> 10 m).** Less than 0.5 m change to median lake level , less than 10% change in mean annual lake level fluctuation and patterns of lake level seasonality (relative summer vs. winter levels) remain unchanged from the natural state.
	Shallow lakes (≤ 10 m). Less than 10% change in median lake level; less than 10% change in mean annual lake level fluctuation and patterns of lake level seasonality (relative summer vs. winter levels) remain unchanged from the natural state.
Medium risk	**Deep lakes (> 10 m).** Between 0.5 and 1.5 m change to median lake level; and less than 20% change in mean annual lake level fluctuation and, patterns of lake level seasonality (relative summer vs. winter levels) show a reverse from the natural state.
	Shallow lakes (≤ 10 m). Between 10 and 20% change in median lake level and annual lake level fluctuation ; and patterns of lake level seasonality (relative summer vs. winter levels) show a reverse from the natural state.
High risk	**Deep lakes (> 10 m).** Greater than 1.5 m change to median lake level; greater than 20% change in mean annual lake level fluctuation, and patterns of lake level seasonality (relative summer vs. winter levels) show a reverse from the natural state.
	Shallow lakes (≤ 10 m). Greater than 20% change in median lake level; greater than 20% change in mean annual lake level fluctuation; and patterns of lake level seasonality (relative summer vs. winter levels) show a reverse from the natural state.

3.2.2 Wetlands

The distribution and occurrence of healthy wetlands varies with size and depth and connectivity to other hydrological systems. The risk of changing lake levels decreasing available habitat or adversely affecting communities depends on the depth and the bathymetry and the dominant species present. The risks of deleterious effects are greater in shallower than in deep-water wetlands.

Table 3.4: ***Descriptors of hydrological change for wetlands.***

Risks under a potential change to flow regime for wetlands, may be defined as follows:	
Low risk	Less than 0.2 m change in median water level; and, patterns of water level seasonality (summer vs. winter levels) remain unchanged from the natural state.
Medium risk	Greater than 0.2 m and less than 0.3 m change to median water level; and patterns of water level seasonality show a reverse from the natural state.
High risk	Greater than 0.3 m change to median water level; and patterns of water level seasonality show a reverse from the natural state.

The effect of changing inflows and/or outflows and therefore changing levels depends not only on the magnitude of change, but also on the periodicity (hydroperiod) and duration of the levels.

3.3 Which Method? Decision-making Framework

3.3.1 Principles for Selecting Methods

a. Lakes

The distribution and occurrence of healthy lake littoral habitats and communities varies with lake size, depth and water clarity. The risk that changing lake levels decrease available habitat or adversely affect communities depends on the lake bed profile (bathymetry), substrate type, water clarity, wave action as well as size and depth. The risks of deleterious effects are greater in shallower systems than in deep water bodies. Within a lake level range impacts arise from changing seasonality in levels and the proportion of time spent at different levels (level duration). Level duration profiles graphically and quantitatively demonstrate the lake level regime (Henderson and Clement 1995; ECNZ 1995; Genesis Power Limited 2000). However there is no easy way to use these in a generic rule-based format as they are generally calculated from absolute altitude. It may be possible to convert these to a relative level base on variance from a mean (or median lake level). This needs to be explored. We recommend that work be commissioned to provide scientific justification for this categorisation and provide an equivalent of mean annual low flow in rivers (and other flow statistics) based on level duration curves.

The decision as to which method to apply for assessing the ecological level requirements of a particular lake depends firstly on the significance of the value to be managed (Section 3.1.2a) and secondly on the potential for hydrological alteration (Section 3.2.1). This framework is presented in Table 3.5: one or more of the methods listed within each cell should be used to assess ecological flow and level requirements for the relevant degrees of hydrological alteration and significance of instream values. In situations with high lake values, two or more methods from each cell should be used, because the risks to ecology of making an incorrect ecological flow decision are greater.

The methods within each table cell are not listed in hierarchical order and the choice of method(s) depends upon the perceived ecological problem affected by the flow regime. For example, if the stability of banks due to wave action, and subsequent effects of turbidity were the perceived major ecological problem of a proposed drawdown in level, then there would be little sense in using a hydrodynamic water quality model. In contrast, if the potential hydrological alteration had the potential to change hydrodynamic processes within the lake, thereby affecting phytoplankton production or oxygen depletion, then a hydrodynamic water quality model should be used where the potential degree of hydrological alteration justified it.

Table 3.5: Methods used in the assessment of ecological flow and water level requirements for degrees of hydrological alteration and significance of lake values.

Degree of hydrological alteration	**Lakes: Significance of values**		
	Low	***Medium***	***High***
Low	Historical time series analysis Expert panel	Historical time series analysis Expert panel	Habitat analysis in drawdown zone Water balance models Species-environment models Residence time vs. water quality modelling
Medium	Historical time series analysis Expert panel	Habitat analysis in drawdown zone Water balance models Species-environment models Residence time vs. water quality modelling	Bank stability and geomorphology analysis Wave action assessment Water level and ramping rates Water clarity assessments Temperature modelling Processes-based water quality models Groundwater/surface water interaction
High	Habitat analysis in drawdown zone Water balance models Species-environment models Residence time vs. water quality modelling	Bank stability and geomorphology analysis Wave action assessment Water level and ramping rates Water clarity assessments Temperature modelling Processes-based water quality models Groundwater/surface water interaction	Bank stability and geomorphology analysis Wave action assessment Water level and ramping rates Water clarity assessments Temperature modelling Processes-based water quality models Groundwater/surface water interaction Hydrodynamic water quality models

b. Wetlands

The distribution and occurrence of healthy wetlands varies with size and depth and connectivity to other hydrological systems. Changes in flow regime are likely to have their greatest effect on wetlands through effects on water level. Falling water tables dry and oxidise soil, leading to peat shrinkage, weed and predator invasion, and loss of specialised wetland flora and fauna. Raising water levels stress wetland plants and decrease plant species diversity, and increase sedimentation and eutrophication. The potential change of the flow regime is therefore defined in terms of its effects on the median annual water table of the site. There may be shorter-term effects on the hydroperiod (eg, changes in seasonality) that can also impact on wetland values.

The risk that changing wetland levels decrease available habitat or adversely affect communities depends on the depth and the bathymetry and the dominant species present. The risks of deleterious effects are greater in shallower than in deep-water wetlands.

The decision as to which method to apply to assess the ecological level requirements of a particular wetland depends firstly on the significance of the value to be managed (Section 3.1.2b) and secondly on the potential for hydrological alteration (Section 3.2.2). This framework is presented in Table 3.6 and one or more of the methods listed within each cell should be used to assess ecological flow and level requirements for the given combination of degrees of hydrological alteration and significance of wetland values. In situations with high wetland value, two or more methods from each cell should be used, because the risks to ecology of making an incorrect ecological flow decision are greater.

The methods within each table cell are not listed in hierarchical order and the choice of method(s) depends upon the perceived ecological problem affected by the flow regime. For example if connectivity with ground or surface waters was a critical factor, then this should be one of the assessment methods used for high-value wetlands; yet such assessment may not be necessary if the wetland is perched without hydrological connections.

c. Decision pathway to setting ecological water levels

The decision pathway that should be followed to set ecological water levels in lakes or wetlands is as follows:

i. identify the spatial boundaries of the lake/wetland, and the impacted area (depth range over which water regime will change)

ii. identify the specific values that will be affected in the impacted area

iii. quantify the range and seasonal timing of change in water level in the affected area

iv. select appropriate methods from the values/change matrix in Tables 3.5 and 3.6

v. quantify the change in hydroperiod consistent with requirements for maintaining values

Table 3.6: Methods used in the assessment of ecological flow and water level requirements for degrees of hydrological alteration and significance of wetland values.

Degree of hydrological alteration	Wetlands : Significance of values		
	Low	***Medium***	***High***
Low ***(< 20 cm change)***	Historical water level records Expert panel Remote delineation of site and catchment Wetland record sheet (MfE methodology)	Historical water level records Expert panel Remote delineation of site and catchment Wetland record sheet (MfE methodology)	Detailed local delineation Wetland hydrological condition assessment and model change (MfE methodology) Species-environment models Habitat assessment Water quality modelling
Medium ***(20–30 cm change)***	Historical water level records Expert panel Remote delineation of site and catchment Wetland record sheet (MfE methodology)	Detailed local delineation Wetland hydrological condition assessment and model change (MfE methodology) Species-environment models Habitat assessment Water quality modelling	Full ecohydrological assessment Groundwater / surface water interaction Process-based water quality models Microtopographic survey
High ***(> 30 cm change)***	Detailed local delineation Wetland hydrological condition assessment and model change (MfE methodology) Species-environment models Habitat assessment Water quality modelling	Full ecohydrological assessment Groundwater / surface water interaction Process-based water quality models Microtopographic survey	Full ecohydrological assessment Groundwater / surface water interaction Process-based water quality models Microtopographic survey

vi. if methods cannot give clear solutions, make 'best information' decision on change in hydroperiod that:

- is suitably conservative
- represents conditions of historical water levels in the system (eg, median historical water levels)
- represents time-varying conditions of water level fluctuations (ie, hydroperiod)
- considers long-term requirements of species for high and low levels (eg, for life cycle passage and recruitment events), including connectivity requirements for fish passage)
- maintains ecological flows in connected surface waters and groundwaters
- includes spatial and temporal limits on level variability.

3.4 Summary of Methods

The following table summarises the methods shown in the decision-making framework, with their advantages and disadvantages.

Table 3.7: Summary list of methods for lakes and wetlands from the decision-making framework (note corresponding shading in cells of Tables 3.5 and 3.6), with pros and cons for use.

Lakes and wetlands		Description	Pros	Cons
1.	Historical time series analysis	Absolute recorded/estimated inflows/outflows, capturing seasonal variability. In the case of wetlands, delineation is required.	Quick and easy, uses existing data, available data, allows variability without going to detailed level of analysis.	Assumes a linear relationship between inflows and lake/wetland habitat; inconsistencies in estimating flow data; difficult to apply in un-gauged systems without accurate models; natural mistrust of method due to being too simple, doesn't target the needs of specific values. Being poorly applied in high value/high change contexts.
2.	Expert panel	Independently appointed panel of experts to advise.	Quick, cheap, has credibility (dependent on experts), can help overcome mistrust if well managed.	Not predictive; can't accurately determine how character of lake/wetland changes with inflows/outflows and levels; can be used as a political tool; implied consensus can lead to poor ecological outcomes.
3.	Site and catchment mapping	Applicable to wetlands rather than lakes. Ground mapping on existing topographic or landcover/soil maps, and or the use of aerial photographs (real or infra-red).	Spatial data, and photographic images are often easy to obtain. Spatial relationships can be inferred and mapped. Department of Conservation is currently developing national database.	Limited ability to predict the extent of level change anticipated or the effects of changing levels.
4.	Wetland record sheet (MfE methodology)	Applicable to wetlands rather than lakes. Uses a set of tables to classify wetlands in a wider context (MfE methodology).	Robust initial analysis using only a set of structured tables to complete.	Requires a good knowledge of wetland systems.
5.	Habitat analysis in drawdown zone	Define lake/wetland habitats on basis of substrate and depth, prevailing wind (in case of lakes) and position of inflows; requires both GIS and field observations (divers/boats for lakes and field surveys for wetlands).	Explicit relationships are obtained between water levels and habitats. Enables location of sensitive habitats, and extent of different habitat types and estimate of non-linear changes in habitat extent with depth.	Only moderately predictive if variability of inflows/outflows/levels changes but not the average values.

Lakes and wetlands		Description	Pros	Cons
6.	Species-environment models	More complete/detailed definition relating to biodiversity and structure of these communities. Emphasis on rare species/unusual communities, or communities that influence the rest of the lake/wetland. For fish, account for feeding/spawning migrations.	Explicit relationships are obtained between water levels and communities. Enables location of sensitive communities, and extent of different community types. For fish it involves the identification of critical areas for fish movements between lakes/wetlands and streams.	Predictive ability is dependent on knowledge of community responses to variable water levels. (Constraints on predictive ability only applies if change in variability of flows and levels is proposed.)
7.	Wetland hydrological condition assessment and model change	Applicable to wetlands rather than lakes. An extension of the Wetland Record Sheet above but more detail and has a data requirement for the scoring of wetland condition (MfE methodology).	Robust analysis using both a set of structured tables and existing data to complete a defined scoring system. Some predictive ability but based on expert opinion.	Requires a data set including some biodiversity and physico-chemical parameters and an expert knowledge of wetland systems.
8.	Water balance models	Model that relates lake change in storage (and therefore levels) to inflows / outflows knowing the lake bathymetry.	Can be used dynamically (daily or fewer time steps), spreadsheet approach.	Dependent on quality of inflow/outflow information and unknown groundwater inputs, and estimates of evaporation (note on method).
9.	Residence time vs. water quality modelling	Applies to lakes and any wetlands with significant areas of standing water. Empirical model relating nutrient loads and residence time to algal blooms and water clarity.	Simple method for professional practitioners.	Some information on nutrient loading and lake/wetland water quality. Applicable to residence times > 10 days and < 10 years. Need limnological experience.
10.	Detailed local delineation	Applicable to wetlands rather than lakes. Delineation is the precise mapping of the wetland boundary at different water levels.	Allows detailed identification of communities in wetlands in relation to hydrology to assist with predictions of effects of change.	Requires expertise in wetland ecosystems including knowledge of hydrology, soils and vegetation.
11.	Bank stability and geomorphology analysis	Assessment of erosion potential of lake shorelines and sensitivity to sediment slumping in response to reduced water levels.	Incorporates an effect of changes in lake water level that is known to be important and is already well understood.	Requires thorough engineering expertise to carry out.

Lakes and wetlands		Description	Pros	Cons
12.	Full ecohydrological assessment	Applicable to wetlands rather than lakes.	Standard international method for detailed understanding of wetland species distribution in relation to hydrology.	Requires a high level of expertise to complete accurately.
13.	Microtopographic survey	Applicable to wetlands rather than lakes.	Refines prediction of effects of water level changes to specific species and communities.	Can be expensive, especially for large sites, and requires expert practioners to complete accurately.
14.	Wave action assessment	An important aspect of hydrology affecting shoreline species distributions in lakes.	Considerable evidence and good models already available.	Requires some expertise in modelling.
15.	Water clarity assessments	Optical assessments of lake water in varying field conditions.	Strong relationships between lake levels, wave action, inflows and clarity can be obtained. Predictive.	Requires expert field work and a good knowledge of factors affecting lake water clarity.
16.	Temperature modelling	1D models of lake vertical structure in the open water or, if requires adjacent to inflows to determine effects of inflows.	Well documented in the literature and predictive.	Requires expert in lake systems and in computational modelling.
17.	Groundwater/surface water interaction	Often critical in controlling species distribution through effects on nutrient inputs.	Often critical in controlling species distribution in wetlands.	Requires interdisciplinary teams for accurate and sound implementation.
18.	Hydrodynamic water quality models	1D, 2D, or 3D representation of temperature, (salinity,) currents and mixing used alone when physical structure of water column and the outflows are of concern. May also include dissolved oxygen, nutrients and chlorophyll (algal production/species)	Better able to predict ecosystem response.	Data, computationally intensive, requires hydrodynamic modelling expertise and expert in-lake process understanding.

3.5 Description of Individual Methods

Lake and wetland water levels are determined by the difference between inflows and outflows (including precipitation and evaporation) at any point in time and the starting level. An understanding of basin shape is essential for any calculations of water level. Therefore any changes to inflow or outflow regimes (units as m^3/s) will be reflected in lake or wetland levels. Water levels may be strongly influenced by groundwater inflows. The discussion below centres on levels but recognises that the inflow-outflow regime is central to setting ecological flows or water levels.

3.5.1 Historical Time Series Analysis

Framework for use (Tables 3.5 and 3.6):

Hydrol. alteration	Values L	Values M	Values H
L	✓	✓	
M	✓		
H			

These methods are based on time series records of water levels and are the simplest and easiest to apply. Analysis of historical time series is based on water levels records and uses a statistic to specify maximum and minimum levels. The statistic could be the average level, the length of time a level must be within a specified range, or the length of time a level may exceed the minimum and maximum levels. A minimum of five years' data would be needed to avoid biases due to natural inter-annual variability. There are much longer water level records available for most of the larger lakes in New Zealand.

The aim of historical analyses is to maintain the levels within the historical range, or to avoid the level regime from deviating largely from the natural regime. The underlying assumption is that the ecosystem has adjusted to the levels regime and that changes from this will cause reduction in the biological state (abundance, diversity, etc) proportional to the change in level. It is usually also assumed that the natural ecosystem will only be slightly affected as long as the changes in level are limited and the water body maintains its natural character. It is implicitly assumed that the ecological state cannot improve by changing the natural water level regime.

The most detailed example of application of rules for lake level management in New Zealand has been the development of the lake operational guidelines for Lakes Manapouri and Te Anau (Mark and Kirk 1987). A 35-year record was used to establish the natural range. The operational guidelines define high, main and low operating ranges, and restrict the amount of time the water level is allowed outside the main range. Long time series records were used for the calculation of lake level duration profiles in the analyses of levels in Lake Taupo (ECNZ 1995; Mighty River Power 2001) and Lake Rotoaira (Genesis Power Limited 2000). These profiles record the percentage of time that the lake has different levels and provide an excellent visual and quantitative summary of the extent of level change.

Bathymetric lake maps are important; there is limited ability to predict the extent of level change anticipated or the effects of changing levels without bathymetric data.

The relationship between level changeand biological response in lakes and wetlands is of course not linear. In shallow water, small fluctuations have a much greater effect than the same fluctuations in deeper water (Walker and Coupland 1968). For this reason it is recommended that the historical time series analysis be carried out by an expert panel if the levels will vary by > 10% of the natural regime in summer and > 30% in winter, unless there is a quantified relationship between biological response and levels.

Recommendation: A historical time series analysis is the most useful simple method to apply to problems of assessing flow regime and water level changes on lake and wetland ecological issues. It is recommended that an expert panel carry out the historical time series analysis if the levels will vary the range or the median by more than 10% unless there is a quantified relationship between biological response and levels.

3.5.2 Expert Panel

Framework for use (Tables 3.5 and 3.6):

Hydrol. alteration	Values L	M	H
L	✓	✓	
M	✓		
H			

Expert panels may be used to provide assessments of effects of changes to flows and levels in situations where the lake and wetland values are low to medium and where the available information on the waterbody is sparse. Expert panels usually comprise interested parties as well as 'experts'. Expert panels may inspect the system and consider the suitability of suggested ecological levels or flows. If records are available, the panel will consider the proposed ecological water levels or flows and particularly the timing of these in relation to the combined knowledge of life cycle requirements of the organisms most likely to be affected. The suitability of ecological water levels or flows for many aquatic biota may be assessed by the panel provided panel members have relevant experience. Particularly important is the ability of the panel to recognise the boundaries of the system and understand the non-linear nature of the biological responses to level changes.

The panel may decide ecological water levels or flows. The panel should justify this decision and identify information needs for future decisions on ecological water levels or flows. It is recognised that results will be dependent on the objectivity of the experts.

Recommendation: Expert panels can design hydrological regimes that support existing biological values, prevent erosion, and ensure water quality remains within specified

ranges. It is a quick and cheap method that builds consensus with stakeholders, but its effectiveness is limited by the credibility of the experts, it is not quantitative or objective, and the need for consensus can lead to inaccurate outcomes.

3.5.3 Site and Catchment Mapping

Framework for use (Table 3.6):

Hydrol. alteration	Values L	M	H
L	✓	✓	
M	✓		
H			

This is particularly important for wetlands, which may not have distinct boundaries with catchments. Site and catchment (at least immediate surrounding catchment) mapping is needed to define the approximate boundaries of the wetland, soil types, the approximate water depths, location of open water and channels, and if possible major vegetation types. All of the above determine the values (eg, distribution of flora and fauna) and are affected by water levels and the periodicity of water level change. Mapping can be done by ground-truthing aerial photographs, working with soil and land cover maps (eg, LCDB2 (Terralink International Ltd), or satellite imagery. Landcare Research has now completed a delineation of wetlands across New Zealand for the Department of Conservation, to be available for this purpose.

Recommendation: The identification of spatial extent of communities requiring protection, of connectivity with other water bodies, and of communities likely to be affected by a water level change is critical for wetland assessments. Remote-sensing/GIS-based methods for wetland identification are straightforward and databases for this purpose are now available for New Zealand. They can be applied for cases of low to medium significance and potential change, but should be supplemented with direct site-specific field data in cases of higher value or potential change.

3.5.4 Wetland Record Sheet

Framework for use (Table 3.6):

Hydrol. alteration	Values L	M	H
L	✓	✓	
M	✓		
H			

This method is provided in the Handbook for Assessing Wetland Condition (Clarkson et al. 2003), developed to allow condition assessments for wetland environments in New Zealand, and was based on the major pressures affecting wetland condition (drainage, nutrient enrichment etc). It included a specific section on hydrological modification that is directly relevant to the type of water abstraction that would occur with flow variation. The Handbook covers in detail the types of hydrological damage that occur in wetlands and their effects on wetland values, and how to assess them. The record sheet is a starting point, and hence minimum requirement, for assessing effects of flow modification on wetlands.

The method (Clarkson et al. 2003) requires completion of a table to classify a wetland in a wider context including evaluation of the value of a wetland in terms of type, regional distribution, rarity. The method requires a suitably qualified person with a basic knowledge of wetland systems. A site visit is essential but the requirement for data is minimal as long as the assessment is carried out by a qualified analyst.

The Handbook is linked to other tools developed for wetland management in New Zealand, including the wetland classification scheme (Johnson and Gerbeaux 2004), and the forthcoming Department of Conservation's lake and wetland prioritisation (Waterbodies of National Importance, WONI) that will provide a multi-factor prioritisation of wetlands in terms of conservation value (contribution to rare habitats and species).

The Wetland Record Sheet component of the methodology is a simple summary of the condition, components and pressures on a site that can be easily used after relatively little training, and suitable for the 'low to medium' level. Other more detailed components of the methodology are discussed below.

Recommendation: The Wetland Record Sheet, part of the Ministry for the Environment condition methodology, is a simple method for the identification of existing hydrological condition and other important issues in wetlands to provide context for assessing effects of proposed change in hydroperiod. It is suitable for cases of low to medium value and potential change of flow regime, but should be supplemented by other information used in the MfE protocol and other methods in cases of higher value and potential change.

3.5.5 Habitat Analysis in Drawdown Zone

Framework for use (Tables 3.5 and 3.6):

Hydrol. alteration	Values L	M	H
L			✓
M		✓	
H	✓		

Habitats refer to the substrate type and water depth. Examples in lakes are rocky platforms, rocky reefs, cliffs, sand, gravel and mud in relation to water depth. In large lakes, wave action usually results in the formation of wave-cut platforms below water level and beaches at and above water level, where substrate type permits (Mosley 2004). In wetlands substrates may be peaty or mineral, with a variety of particle sizes, and water depths may range from areas that are only occasionally flooded to areas that are permanently submerged.

The proportion of lakeshore occupied by various habitat types and their depth distributions need to be quantified. Where sensitive habitats occur, their proportional loss or gain at various lake levels is calculated. Because lake edges are seldom uniformly sloping, the first stage in an analysis of habitats is from a hypsographic curve on which lake levels can be plotted. Wave-cut platforms near the lake surface imply that significant loss of lake area and hence lake littoral communities may occur for relatively small changes in lake level. Water level variation along cliff faces, in contrast, may have little biological impact.

Similar issues arise in wetlands, where community composition and species diversity of plants as well as animals vary across sites with different substrates and water depths.

Many of the values in Tables 3.1 and 3.2 will be impacted by water level change if this change coincides with areas of gentle slope thereby removing significant areas of habitat. Calculations may be based on:

- rarity of habitat type impacted by the level change; for rare habitats any loss may be considered unacceptable
- proportion of habitat type impacted by the level change; for common habitats less than 20% loss may be considered 'Low impact' and more than 30% loss may be considered 'High impact'
- ability of the biological communities in the habitat to migrate up or down with water level. Some plants that provide habitat structure are highly adaptable to water depth variations (eg, characean algae) and some, mainly those that require exposure for flowering at some stage in the life cycle and therefore live only in water depths < 1 m are highly intolerant of deep water. Many shallow-water plants may survive long (weeks') exposure to air in winter, but are intolerant of short (days') exposure in summer.

This assessment needs to be made by an experienced practitioner.

Recommendation. Habitat analysis is applicable where the values are Medium to High. Required method when explicit relationships between depth and habitat area are needed. Method provides proportional change to the depth distribution of wetland and lake littoral habitats resulting from the water level change. For rare habitats no loss is acceptable, for common habitats < 20% loss is Low impact, 20–30% is Medium impact and > 30% High impact.

3.5.6 Species-environment Models

Framework for use (Tables 3.5 and 3.6):

Hydrol. alteration	Values L	Values M	Values H
L			✓
M		✓	
H	✓		

Lake and wetland communities can be categorised into four major groups that respond (often in concert) to changes in water levels. These are aquatic macrophytes and periphyton, aquatic macroinvertebrates, fish, and birds.

Aquatic macrophytes are fundamentally important in lake-edge structure and in defining the characteristics of wetlands. These may be submerged, emergent or free floating (de Winton and Schwarz 2004). Assessments in relation to water level change require two stages.

The first stage in a macrophyte assessment is a biodiversity analysis, including:

- number of species, particularly native species
- identification of rare species
- analysis in the depths that will be impacted by an ecological level or flow regime.

The second stage is an assessment of the depth distribution of the communities within the water body. Methods are summarised by de Winton and Schwarz (2004). The LakeSPI method for water quality assessment (Clayton et al. 2002) may also provide the necessary information. With this combined information a proportional loss (gain) of macrophyte species and of macrophyte-dominated communities can be calculated by overlaying the proposed water level regime on the existing regime using either direct transects and/or the hypsographic curve as outlined in Section 3.5.5 An expert judgement is then required as to whether this loss is significant: there are no current quantitative methods to define importance and there is the potential for the plants (depending on the species) to adjust by upward or downward migration to new levels.

Macroinvertebrates occupy the middle levels of aquatic food webs. Their numbers, distributions and life cycles are often critically dependent on water levels, particularly in shallow water where drying may occur with water level changes.

The wetlands and the littoral zone in most lakes include a variety of habitats, thus necessitating a large number of replicate samples and sites to encompass spatial variability. Additionally, habitats within the lake littoral zone can be extremely diverse (eg, macrophytes, boulders, fine sediment), requiring the use of several sampling methods. The following four methods are recommended by Kelly and McDowall (2004) for macroinvertebrate community assessments in lakes:

- sweep netting
- benthic grabs
- coring
- Hess/Surber sampler.

The pros and cons of the use of these are given by Kelly and McDowall (2004). There are other methods in the literature (eg, detailed diver assisted analyses) but they are generally more time-consuming and will be needed only for very detailed work if warranted. Sampling needs to be conducted to obtain a relationship of benthic macroinvertebrates to water depth and then the same process is applied as for macrophytes above.

Fish communities that will be affected mostly by water level variations are those that utilise or inhabit the littoral zone of lakes. For example, bullies may use plants in the littoral zone as spawning sites, and fish in spawning streams may be affected by variations in lake levels where the spawning stream enters a lake across a delta.

Quantitative assessments of littoral fish communities include:

- seining
- fyke netting
- fish trapping
- gill netting
- electric fishing.

For details of the methods see Kelly and McDowall (2004) and references therein.

There are no native herbivorous freshwater fish in New Zealand so an initial analysis of changes to the water level regime can be obtained from the effects on macrophytes where these provide habitat for fish, and macroinvertebrates where these constitute the majority of fish food.

Quantitative ***bird*** counts in lakes and wetlands may be made by a number of techniques, but variability in numbers due to regular migrations between water bodies in a given area mean that good assessments require at least seasonal and preferably intra-seasonal counts.

The assessments need to be made for species directly associated with habitats that may change as a result of water level variability, eg, species directly dependent on the littoral zone for food or moulting and breeding shelter such as ducks, swans and pukeko (Williams 2004). Some shallow water bodies attract wading birds, and spectacular seasonal changes in abundance can be found where migrant species utilise lake margins (eg, Lake Wairarapa: Williams 2004).

Recommendation: Species-environment models are applicable where the waterbody values are Medium to High. Required method when explicit relationships between depth and community area or numbers and biodiversity are needed. Methods provide information on proportional change to the macrophyte / periphyton, macroinvertebrate, fish and bird communities as a result of water level change. The assessment of impacts of this change on lake values is best done by expert practitioners as there are no quantitative indices that can be used for the importance of the calculated changes.

3.5.7 Wetland Hydrological Condition Assessment and Model Change

Framework for use (Tables 3.5 and 3.6):

Hydrol. alteration	Values L	M	H
L			✓
M		✓	
H	✓		

The method is described in the Ministry for the Environment's 'Handbook for Monitoring Wetland Condition' (Clarkson et al. 2003). It is an extension of the Wetland Record Sheet (Section 3.5.4) supplemented by a data set from the wetland, that includes details of plant species presence and height as well as physical and chemical parameters measured in the field or from laboratory analyses for a number of plots in the wetland. This provides a more robust base for scoring wetland condition, and defines the factors controlling the habitat for biota more precisely. Wetland scores are assigned to a set of indicators using a systematic comparison and evaluation process based on expert knowledge.

The methodology (Clarkson et al. 2003) can be used by any wetland manager after some training from wetland experts.

Recommendation: The Ministry for the Environment condition assessment method recognises hydrological modification as the most important driver of wetland condition; the information provided by the method is suitable for assessing effects of hydrological change when the values are Medium to High. Methods provide information on the vegetation and habitat structure for other organisms as well as hydrology, water and soil chemistry, all of which are important aspects to consider when assessing the effect of change in flow and water level. The assessments can be carried out by any resource managers, provided they have had some training in the application by qualified wetland experts.

3.5.8 Water Balance Models

Framework for use (Tables 3.5 and 3.6):

Hydrol. alteration	Values L	M	H
L			✓
M		✓	
H	✓		

Water balance models that relate the change in lake or wetland storage and therefore levels to inflows and outflows are relatively straightforward to compute. With knowledge of bathymetry, these can be predictive. Water balance models are therefore the next step from habitat or species-environment models as these can provide the predictive component for lake levels on which the habitat or species-environment models can be made.

The method therefore requires both hypsographic information and a time series of inflows and outflows including precipitation and evaporation. Running a simple inflow-outflow model requires keeping a balance between hydraulic inputs and outputs, and how any imbalance causes lake levels to change. It can be carried out over a wide range of timesteps by a simple spreadsheet approach.

Un-gauged rivers and groundwaters entering lakes and wetlands are a potential difficulty. In some lakes and wetlands, evaporation may be significant (usually when surface area to volume ratio is high and in dry windy conditions (eg, Lake Ellesmere). Measurement of evaporation relies only on indirect methods, eg, by pans or energy balance models.

Changing the inflow-outflow-lake level regime will also change the lake residence time with considerable potential flow-on effects to lake ecosystems as discussed in Section 3.5.9.

Recommendation: Water balance models are applicable when the values are medium to high, where lake or wetland inflows are known and where reasonably robust predictions are required for levels as a result of changes to inflows and / or outflows.

3.5.9 Residence Time – Water Quality Modelling

Framework for use (Tables 3.5 and 3.6):

Hydrol. alteration	Values L	M	H
L			✓
M		✓	
H	✓		

The inflow-outflow regime for a given lake or wetland may affect water level and residence time which in turn affects water quality. Simple empirically derived regression models can be used to estimate long-term (or equilibrium) nutrient and chlorophyll concentrations from values of nutrient loadings and lake residence time. Application of these models should be a first step in an assessment of effects on water quality of altering inflow-outflow regimes; it requires knowledge of nutrient loading rates and existing residence times of a lake (Ryding and Rast 1989). The models provide information and predictions on in-lake nutrient concentrations with changing inflows and out flows. Other empirical relationships have been derived relating chlorophyll to in-lake nutrients, as shown in the New Zealand examples in Pridmore et al. (1985).

Consideration of residence time in water quality is not applicable when the residence time is less than approximately 10 days: the system can then be considered a riverine environment. In terms of residence time considerations alone, a rule of thumb might be that if the change in residence time is less than 10%, then residence time is not a useful parameter.

The models, although relatively simple empirical approaches, require some information on nutrient loading and lake water quality. They are applicable when residence times are > 10 days and < 10 years. Experience in limnology is required to develop and apply these models although they are available in the literature (eg, Ryding and Rast 1989).

Recommendation: Residence time models are simple to use, have moderate predictive power and are applicable when there is concern that the water quality may be affected by changing inflows and outflows and where values are Medium to High. They should be used when residence times are greater than 10 days and less than 10 years.

3.5.10 Detailed Local Wetland Delineation

Framework for use (Tables 3.5 and 3.6):

Hydrol. alteration	Values L	M	H
L			✓
M		✓	
H	✓		

Section 3.5.3 above described how remote delineation of wetland environments, using mapping and GIS techniques, is an important step in any assessment of effects of flow change on wetlands. For wetlands with medium to high values, or where the potential change to the water level is medium to high, the boundaries of the wetland environment with the terrestrial catchment are usually an important issue; this often requires a more precise local on-site delineation than is possible from remote techniques. This is because the significance of the wetland community and its susceptibility to hydrological modification are often greatest near the margins (margins are often species-rich and rare communities).

Methods for on-site wetland delineation were initially developed by the United States Army Corps of Engineers, and allow the nature and extent of the wetland and its constituent communities to be determined quantitatively. The methods involve a range of hydrological, soil property and vegetation composition techniques, and can be applied to data collected using the methods described in Sections 3.5.4 and 3.5.7. In order to be applied objectively, only qualified wetland experts can carry out this method. The basic principles are described in American publications such as Tiner (1999); and many of these methods have been modified for New Zealand conditions by local wetland practitioners.

Recommendation: Detailed local delineation is essential for identification of sensitive wetland communities near the terrestrial-wetland boundary and for predicting effects of hydrological change on different communities within a wetland, and should be applied where values are Medium to High. They should only be carried out by qualified wetland experts.

3.5.11 Bank Stability and Geomorphology

Framework for use (Table 3.5):

Hydrol. alteration	Values L	Values M	Values H
L			
M			✓
H		✓	✓

One of the major effects of change in water level in lakes is impairment of lakeshore stability, resulting in shoreline collapse and loss of beach sediments. This problem has received considerable attention in New Zealand, as collapses of Lake Manapouri shorelines in 1972 were an important driver behind the lake operational guidelines for Lakes Manapouri and Te Anau. Geomorphological methods for determining whether a change in flow regime and lake level is likely to cause such problems are well established, but are complex and must be carried out by qualified and experienced engineers. Kirk and Henriques (1986) and Kirk et al. (2000) provided detailed examples of how to carry out these methods.

Recommendation: Changes in bank stability and erosion are well recognised as an important problem in hydrological management of lakes and must be considered where values and potential changes to flow regime are Medium to High or High. Such assessments require expert geomorphologists; there have been many previous examples of such assessments in New Zealand and the methods are well-developed.

3.5.12 Full Ecohydrological Assessment

Framework for use (Table 3.6):

Hydrol. alteration	Values L	M	H
L			
M			✓
H		✓	✓

Given that the extent of wetlands is so reduced in New Zealand, any proposed hydrological alteration that has potential to affect a wetland of medium to high value would need to be assessed with the greatest caution; and with thorough, objective methods for predicting effects of change in response to lowering or raising of water tables. Detailed ecohydrological assessments, which model the distribution and productivity of natural communities in relation to hydroperiod, are the most robust approach. Yet they have not often been applied because of their high cost and need for a high level of scientific expertise. Ecohydrology is an interdisciplinary approach in which hydrologists, soil scientists, and biologists use hydrological data and species distributions to identify gradients and patterns in hydrology-species responses, recognising that even small differences in hydrology can have large effects on biological values when water levels pass critical values. The methodology involves monitoring networks of hydrological instrumentation (dipwells, piezometers and capacitance probes) placed along ecological gradients to characterise hydrology-species composition relationships, and make predictive models for effects of change in hydroperiod. Browne and Campbell (2005) give a recent New Zealand example of a detailed ecohydrological study for wetland management purposes. This approach clearly requires a high level of expertise and should always involve qualified wetland experts.

Recommendation: The ecohydrological approach is the most robust, internationally used method for understanding relationships between hydrology and ecological values in wetlands, and should be applied in all cases of Medium to High and High wetland value or potential for change. It requires a high degree of wetland expertise.

3.5.13 Microtopographic Survey

Framework for use (Table 3.6):

Hydrol. alteration	Values L	M	H
L			
M			✓
H		✓	✓

Wetland communities, and therefore wetland structure and function, are critically dependent on small (often cm scale) changes in topography that may mean the presence or absence of standing water, channelised flows and aerated versus non-aerated soils. Microtopography is therefore a key factor related to hydrology that promotes the development of vegetative structure and composition, and biogeochemical functions. Microtopography requires accurate field mapping and survey. If a full wetland microtopographic map cannot be produced for cost or logistic reasons, then survey cross-sections need to be chosen that best reflect the water level issues that face the wetland on a case-by-case basis. Microtopographic work is usually accompanied by detailed soil profiling. Soil profiles are carried out by coring or, if the soils are relatively dry at the time of study, soil pits. Alteration of wetland levels in peat soil-dominated areas may have a significant effect. Drying of peat-dominated soils invariably results in soil shrinkage and general lowering of soil profiles around the affected wetland area (eg, Lake Poukawa in Hawkes Bay).

Microtopography involves transect elevation measurements using survey equipment. Several organisations in New Zealand have expertise in microtopography. Rapson et al. (2006) provide a recent New Zealand example showing the importance of microtopography and how to design a microtopographic analysis in relation to a hydrological gradient.

Recommendation: Microtopography is a critical component controlling patterns of species diversity and biodiversity value in wetlands in relation to hydrology. Microtopographic surveys should form part of assessments of potential changes in flow and water level in wetlands with Medium to High value.

3.5.14 Wave Action Assessment

Framework for use (Table 3.5):

Hydrol. alteration	Values L	M	H
L			
M			✓
H		✓	✓

Wave action assessments are required when lake levels change to the extent that new shorelines are formed. These assessments are often closely linked with bank stability and geomorphology (Section 3.5.11).

Data requirements include an outline of the lake shoreline, bottom bathymetry, and information on wind speed and direction. Wave action assessments have been carried out for example on Lake Taupo to simulate effects of lake level changes on shoreline erosion (Hicks et al. 2000), and this method can also be applied to assess the effects of waves on littoral zone vegetation and biological communities. The method involves application of a wave hindcasting model coupled with a wave refraction model. The underlying science and methodology may be found in the Coastal Engineering Manual (http://www.veritechinc.net/products/cem/index.php).

Recommendation: Applicable to lakes when lake level changes are likely to impact on shorelines either by erosion or accretion when waterbody values are Medium or High. Methods for wave analysis on shorelines are available but expert practitioners are required for the analysis and interpretation. The method also allows predictions to be made to habitat and species-environment models (Sections 3.5.5, 3.5.6) where these are likely to be affected by wave action.

3.5.15 Water Clarity Assessments

Framework for use (Table 3.5):

Hydrol. alteration	Values L	M	H
L			
M			✓
H		✓	✓

Water clarity is influenced by lake levels in two ways.

First, lake levels influence sediment re-suspension; suspended sediment effects are most important in lakes where the shorelines are gently sloping and/or where much of the lake bed is influenced by wave action. Changing lake levels in deep lakes with steep sloping shores will have a relatively small influence on lake clarity (and hence ecosystem effects) when compared with lake level changes in shallow lakes with gently sloping shores. Changing inflows may alter the sediment loading and hence clarity of lakes, as is the situation for Lake Manapouri and Lake Coleridge where diversions of sediment-rich water are controlled to minimise inputs to the lakes.

Second, lake levels influence lake algal growth. The effect of lake level change on water clarity via changes in algal (phytoplankton) concentrations is indirect. If the change in water level regime alters the primary drivers of phytoplankton dynamics, nutrients temperature and light, then it will influence phytoplankton biomass and hence water clarity.

The method involves considerable field work to measure light penetration both inshore and offshore in different wind and wave conditions, and at different times to derive relationships between clarity and sediment and phytoplankton concentrations. The latter may be predicted from their residence time models (Section 3.5.9) and wave action assessments (Section 3.5.14).

An assessment of lake level influences of this nature requires considerable knowledge of lake ecosystem processes and a good knowledge of the relationships of sediment and phytoplankton biomass and clarity relationships which vary from lake to lake (Hamilton et al. 2004).

Recommendation: Clarity assessments should only be made if the lake values are Medium to High, if clarity is a recognised water quality value for the waterbody, and if it is clearly under threat from the changing inflows and / or lake levels. The method requires field work, detailed measurements of clarity and associated factors, and an expert knowledge of lake ecosystems.

3.5.16 Temperature Modelling

Framework for use (Table 3.5):

Hydrol. alteration	Values L	Values M	Values H
L			
M			✓
H		✓	✓

The temperature of a river inflow determines the amount of heat that the river delivers to, or removes from, a lake; and also the depth in the lake to which the inflowing water will sink. This latter effect is generally of greater concern, because of the substances that may be transported by the inflow (oxygen, nutrients, suspended particulate matter); and because of the control that thermal stratification exerts on vertical mixing and transport of these substances within the lake.

Application of a 1D hydrodynamic model that simulates vertical thermal structure and predicts insertion depths for inflows is helpful to understand the dynamics of river-lake interaction.

Outflow dynamics can influence in-lake thermal structure in deep artificial reservoirs with offtakes at depth, but this will not be considered here.

An example for the inflows from the Tongariro Power Development Scheme into Lake Taupo using a 1D model is given in Spigel et al. (2005) and for Lake Rotoiti using a 3-D model is given in Stephens (2004).

Recommendation: Temperature modelling should only be used when there is considerable existing knowledge of lake water column structure and where there is a clear threat that the changing inflows will detrimentally alter this. Lake temperature models are well documented and are predictive but are complex and need to be run by experienced lake experts.

3.5.17 Groundwater / Surface Water Interaction

Framework for use (Tables 3.5 and 3.6):

Hydrol. alteration	Values L	M	H
L			
M			✓
H		✓	✓

Groundwater inputs are an important component of the hydrology of wetlands and shallow lakes, and changes in groundwater input potentially have significant effects on nutrient inputs and hence ecological character. Rates of groundwater discharge are known to be important in sustaining productivity of the littoral vegetation of lakes and of wetland communities. Groundwater discharges from catchments into wetlands and lakes can be estimated from methods such as piezometer clusters, Darcy calculations and salt balances. The groundwater section of this document has further details of methods for estimating groundwater discharge.

Recommendation: Groundwater inputs are an ecologically significant component of wetlands and shallow lakes; effects of hydrological alterations on groundwater inputs should be considered in all cases of Medium to High values and potential change to flow regime.

3.5.18 Hydrodynamic Water Quality Models

Framework for use (Table 3.5):

Hydrol. alteration	Values L	M	H
L			
M			✓
H		✓	✓

A comprehensive process-based lake model may be used if a risk of adverse effects is identified and the results from empirical residence time modelling, clarity and temperature modelling (Sections 3.5.9, 3.5.15, 3.5.16) does not provide sufficient information to assess ecological water levels and flows. Such modelling requires extensive supporting data including climate, inflows and the in-lake conditions.

The first stage is the implementation of a lake hydrodynamic model. Such a model provides necessary background information on thermal structure, mixing, and water movements that can be used to underpin water quality and aquatic ecosystem models. A suite of hydrodynamic models for different purposes and lake types are available (see Hamilton (1999) for a discussion of models and their applications).

In a combined hydrodynamic-water quality ecosystem model, the degree of spatial and temporal resolution provided by the hydrodynamic model controls the overall resolution of the simulation. Hence a 1D thermal stratification model will limit water quality predictions to profiles that represent lake-wide averages. If 2D or 3D effects are of interest or cannot be ignored (as in very large shallow lakes) then 2D or 3D hydrodynamic models must be used in combination with a water quality model.

Following calibration and verification of a hydrodynamic model, the second stage is the coupling of this to a lake water quality model. It is probably best to run the hydrodynamic and the water quality models as a coupled system. To capture any interactions between the physics, chemistry and biology, it is also possible to run the models in an uncoupled mode: output on physical factors from the hydrodynamic model are then used to control the movement and mixing of water quality components in later, separate water quality model runs. Process-based water quality models simulate interactions between lake inflow-outflow regime, lake physical dynamics, water quality and biological components.

An example of such a modelling exercise in relation to changing inflows is given in Hamilton et al. (2005) for Lake Rotoiti, simulating effects of a flow diversion into and out of the lake.

Recommendation: Where lake values are High and the potential change to the whole lake ecosystem from the proposed flow regime is High and when there is considerable existing information on the lake, its catchment and its inflows and outflows – then hydrodynamic water quality modelling may be appropriate. These models are complex and need to be run by lake experts. Some of these models are available on the web but most are commercial.

4 Groundwaters

The purpose of this section is to describe technical methods for assessing ecological flow and water level requirements for groundwater systems. Such requirements are set in the context of environmental, social, cultural and economic values of a water body (aquifer). Our approach concentrates on the aspects of groundwater systems related to ecological values (in the groundwater system and connected surface water systems) and physical properties of the aquifers such as structure and water quality. It does not include an assessment of wider economic and social factors such as reliability to water users.

For groundwaters, an ecological flow regime may include an allocation limit, water level or pressure limits, or other measures to ensure management objectives (such as adequate surface water flows or prevention of salt water intrusion) are met. Therefore, the ecological flow or water level regime includes allocation limits but often includes other measures as well.

The ecological flow regime in groundwater may vary in different circumstances. For example the ecological flow regime in groundwater may consist of one of a:

- simple groundwater allocation limit
- groundwater level limit with a groundwater allocation limit
- minimum flow restriction in a stream and a groundwater allocation limit.

The process of setting ecological flows and water levels in groundwater systems involves three steps:

1. assessment of resource values and their relative significance
2. assessment of the degree of hydrological alteration that could be expected from groundwater allocation
3. identification of an appropriate method, or methods, for the assessment of ecological flow/water level requirements.

The process of setting ecological flows and water levels in groundwater systems should consider uncertainty and the unknowns associated with groundwater systems.

Therefore the approach for groundwater aims at:

- a conservative approach to method identification
- applying, at least, the basic groundwater assessment method (conceptual model / simple water balance) for groundwater systems
- applying more complex methods for higher resource values and higher degrees of hydrological alteration
- adopting a ‘cumulative approach’ to methods application.

Typically, knowledge of groundwater systems is less certain than knowledge of surface waters. Therefore the approach for groundwater differs slightly from the approach for rivers, lakes and wetlands. A ‘cumulative approach’ to groundwater methods application is used in response to uncertainty and the unknowns associated with groundwater systems. A 'cumulative approach' to methods application follows the typical groundwater investigation process whereby simple models are used to build more complex models.

4.1 Assessment of Resource Values and their Relative Significance

An assessment of resource values is an important part of selecting an ecological flow regime. It establishes the natural water body systems that could be affected and thus the methods of assessment to be used, as well as establishing baseline data for the consideration of environmental effects.

Values may be broadly grouped into:

- aquifer integrity including water use values
- aquifer outflow values
- ecological or water quality values.

There are 'flow-related values' that change in a discernible way as flow changes within aquifers from variations to aquifer recharge, groundwater abstraction, or modifications to aquifer outflows. Table 4.1 lists some groundwater values relevant to management of aquifer systems.

Table 4.1: Some Groundwater Values, or Management Objectives, for Aquifer Systems and Factors to be Considered in Achieving the Management Objectives.

Some groundwater values or management objectives	Some factors that affect ability to achieve objective
Maintaining outflows that sustain surface water	Groundwater head, and gradient
Maintenance of groundwater ecology (flora and fauna)	Groundwater head variation
Controlling land subsidence and aquifer consolidation	Groundwater head
Controlling saltwater intrusion	Groundwater head, and gradient
Maintaining groundwater quality	Point and non-point sources of pollution, groundwater head and groundwater flow
Recharge	Land use, rainfall, evaporation, river use, river flow, river bed condition
Maintain surface water quality	Groundwater head, and gradient, groundwater quality
Groundwater storage	Groundwater recharge, groundwater discharge
Maintain head or pressures	Groundwater recharge, groundwater discharge, groundwater use

It is often not possible to detect change in aquifer conditions as groundwater flows are reduced or the pattern of flows is changed. The inability to detect change arises from the high natural variability and the complexity of aquifer-surface water systems. It is only once springs stop flowing or wells dry up, that it becomes clear that values cannot be sustained. Management approaches need to reflect the associated uncertainty in aquifer response.

4.2 Determination of the Degree of Hydrological Alteration

Natural groundwater flow is altered by groundwater use. Natural groundwater flow is altered at the local scale (eg, pumping of groundwater from a well inducing groundwater flow towards the well) and altered at the regional scale (eg, cumulative groundwater use reducing flows in spring-fed streams). The impacts on natural groundwater flows will depend on the amount of use, the location of use, the timing of use, and aquifer properties.

The setting of ecological flows and water levels controls the amount, the location, and the timing of groundwater use. Groundwater ecological flows or water levels are linked with surface water ecological flows where effects of groundwater use impacts on surface water.

The amount of groundwater allocated by resource consent is typically greater than groundwater use. For example the annualised groundwater allocation is approximately seven times greater than groundwater use in the area between the Ashley and Ashburton rivers in Canterbury (White et al. 2003). This approach is based on the amount of water allocated, rather than the amount used. It is important that over time, allocation approaches change to better reflect actual use and seasonal volumes.

The degree of hydrological alteration of a groundwater system is related to the amount of groundwater allocated. Hydrological alteration of a groundwater system is related to groundwater allocation in three classes:

- low, where the allocation is a small proportion of recharge and therefore ecological effects of groundwater use are likely to be minor
- medium, where the allocation is a moderate proportion of recharge and therefore ecological effects of groundwater use are likely to be moderate
- high, where the allocation is a large proportion of recharge and therefore ecological effects of groundwater use may be significant.

These classes also relate to the security of supply for groundwater users and the conditions of groundwater allocation. For example groundwater users will have high security of supply where the hydrological alteration of a groundwater system is low and resource consents may include minor restrictions on groundwater use. However groundwater users may have less security of supply where the hydrological alteration of a groundwater system is high because the resource consent may include major restrictions on groundwater use. The conditions on groundwater consents are commonly linked to effects of groundwater use, including: local effects such as drawdown in a neighbouring well or induced flows from streams; and regional effects such as cumulative groundwater use reducing flows in spring-fed streams.

Hydrological alteration of a groundwater system is related to groundwater allocation in three classes by the portion of groundwater allocation to recharge from surface water sources, ie, (with percentages rounded):

- low, where allocation is up to 10% of recharge from surface water sources
- medium, where allocation is from 11% to 25% of recharge from surface water sources
- high, where allocation is greater than or equal to 26% of recharge from surface water sources.

Existing groundwater allocation is typically assessed using regional council, or district council, resource consent databases. Surface recharge is typically assessed using methods outlined in Section 4.5. Estimates of median recharge (rather than mean) should be used if recharge estimates are available as time series because median estimates are more conservative than mean estimates.

These figures are based on experience rather than research and are conservative. They recognise some experiences of the effects of groundwater allocation in New Zealand. For example the approach would have the area between the Rakaia River and the Waimakariri River on the Canterbury Plains classed with a 'high' degree of hydrological alteration based on existing groundwater allocation. Allocation is approximately 119% of recharge from surface water sources because: estimated annualised allocation is around 43 m^3/s (White et al. 2003) and estimated groundwater recharge from surface water is around 36 m^3/s (made up of around 24 m^3/s rainfall recharge (White et al. 2003), around 7 m^3/s recharge from the Waimakariri River, and up to 5 m^3/s recharge from the Rakaia River, (Bowden 1983)). Groundwater levels in the area are commonly observed below their long-term average (eg, NIWA 2004) possibly because groundwater use is a significant portion of groundwater recharge.

4.3 Which Method? Decision-making Framework

4.3.1 Principles for Selecting Methods

Groundwater flow, or level, assessment tools are commonly used in the assessment of impacts of groundwater abstraction in New Zealand. These assessment methods are summarised in Section 4.4, and described in detail in Section 4.5.

Table 4.2 outlines the *selection* process of method*s* based on 'resource values and their relative significance' and 'potential degree of hydrological alteration from groundwater allocation'. Resource values, and the degree of hydrological alteration from groundwater allocation, need to be carefully considered in the context of the management objectives outlined in Table 4.1 to evaluate ecological flows/water levels with *appropriate methods*.

Each method (Section 4.5) includes a 'decision pathway to setting ecological flows'. Application of this pathway results in a cumulative application of methods as development pressure increases on a groundwater system. For example the pathway to applying the 'historical levels' method (Section 4.5.2) includes application of the 'conceptual model/simple water balance' method (Section 4.5.1).

The selection process aims to have 'resource values and their relative significance' as the main criteria for *identifying* methods most suitable for ecological flow requirements when the relationship between the potential change to the flow regime *and groundwater allocation* is uncertain (eg, in deep confined aquifer systems where *groundwater* recharge and *groundwater* discharge are not well defined).

Table 4.2: Methods used in the assessment of water level requirements for degrees of hydrological alteration and significance of groundwater values.

Potential degree of hydrological alteration from groundwater allocation	Groundwater: Resource values and their relative significance		
	Low (not sensitive)	**Medium**	**High (extremely sensitive)**
Low (up to 10% of recharge)	Conceptual model /simple water balance Historical levels	Conceptual model /simple water balance Historical levels Expert panel Detailed water balance	Detailed water balance Time series analysis Analytical models Numerical quantity models – steady state Numerical quantity models – transient Numerical quality models – transport
Medium (11–25% of recharge)	Conceptual model / simple water balance Historical levels Expert panel	Detailed water balance Time series analysis Analytical models Numerical quantity models – steady state	Numerical quantity models – steady state Numerical quantity models – transient Numerical quality models – transport Consolidation models
High (over 25% of recharge)	Detailed water balance Time series analysis Analytical models Numerical quantity models – steady state Numerical quantity models – transient Numerical quality models – transport	Numerical quantity models – steady state Numerical quantity models – transient Numerical quality models – transport Consolidation models	Numerical quantity models – steady state Numerical quantity models – transient Numerical quality models – transport Consolidation models

The classification of 'degree of hydrological alteration from groundwater allocation' (Table 4.2) considers groundwater allocation and recharge. This classification aims to:

- provide a consistent approach to setting ecological flows and water levels
- provide an increasing knowledge base for decisions on ecological flows and water levels as development pressure increases.

Method selection should aim at a conservative approach after considering an analysis of uncertainty. For example methods will be in the 'Medium' category for 'potential degree of hydrological alteration from groundwater allocation' where groundwater allocation is 20 ± 5 %) of groundwater recharge, but qualifies for a 'High' classification where groundwater allocation is 20 ± 8 % of recharge. In other words, the degree of uncertainty associated with 'allocation as a percentage of groundwater recharge' needs to be taken into account when determining the appropriate 'hydrological alteration' category. Where doubt exists, users should defer to the highest appropriate category.

Method selection should also consider the surface water methods, where surface water is linked to groundwater.

4.3.2 Decision Pathway to Setting Ecological Flows and Water Levels

The overall process used in application of the approach to a given situation is as follows:

1) identify the groundwater system, boundaries and stresses
2) consider linkages between groundwater and surface water
3) identify the values or management objectives of the system (Table 4.1)
4) set management objectives and criteria significance, based on the values, ie, decide the significance of values and the potential change to the flow regime for groundwater, and for surface water if appropriate
5) identify critical factors in relation to groundwater system allocation and level, and for surface water if appropriate
6) select the potential methods from Table 4.2 based on 'significance of groundwater resource values' and 'degree of hydrological alteration'.

For example potential methods are conceptual model/simple water balance, historical levels and expert panel for the cell in Table 4.2 where the 'significance of groundwater resource values' is medium and 'degree of hydrological alteration' is low.

7) select an appropriate method from the potential methods to set groundwater ecological flows and water levels in the aquifer system based on:
   - the class of problem
   - data availability.

The class of problem may be either groundwater quantity or groundwater quality, or both. For example maintenance of base flow in spring-fed streams is a groundwater quantity problem so a numerical quantity model is a relevant method.

Data availability is a key issue in method selection: a conceptual models/simple water balance can require little data, whereas a credible transient groundwater flow model has a large data requirement. Methods in each cell in Table 4.2 are listed in order of increasing data needs. A method should be chosen that is consistent with available data.

For example the method 'numerical quantity model – steady state' is appropriate where:

- 'significance of groundwater resource values' is high and 'degree of hydrological alteration' is low
- the class of problem is groundwater quantity
- data is available to build a credible model.

8) apply the method, considering critical factors and knowledge requirements, following the 'detailed decision pathway' of each method (Section 4.5), to:

- set a groundwater ecological flow/water level, and/or the groundwater level regime, to protect or manage the relevant objective(s)
- set a surface water ecological flow, if appropriate.

4.4 Summary of Methods

Table 4.3 summarises the methods shown in the decision-making framework (Table 4.2) and their advantages and disadvantages.

Table 4.3: Summary of methods for groundwaters from the decision-making framework (Table 4.2), with advantages and disadvantages for use.

Method	Description	Pros	Cons
Simple water balance	Estimating inflows and outflows	Simple	Uncertainty
Conceptual model	Physical aquifer evaluation / environment (including water flows)	Easy, provide basis for further assessment	Uncertainty, information
Historical levels	Examination of measured levels – trend and seasonality	Simple, directly related to aquifer performance	Data dependent – quantity / quality / spatial
Expert panel	Independently appointed panel of experts to advise	Quick, cheap, has credibility (dependent on experts), can help overcome mistrust if well managed	Not predictive, can't determine how character of aquifer changes with level, can be used as a political tool, implied consensus can lead to poor environmental outcomes
Time series analysis	Statistical analysis of levels to identify system drivers	Relatively easy, can allow testing of scenarios	Need reasonable time series; misinterpretation
Detailed water balance	Quantifying inflows and outflows	Applied to all situations	Subsurface recharge and discharge unknown
Analytical models	Spreadsheet models based on groundwater flow and transport equations	Ease of use, moderate skill level, moderate data requirement	May ignore cumulative effects; requires simple assumptions
Consolidation models	Definition of settlement from depressurisation	Allows quantification	Sophisticated, data requires knowledge of use
Numerical quantity models (steady state)	Iterative-spatial representation of hydrogeology based on flow equations	Allows greater representation of real world	Time, complexity, data, misuse
Numerical quantity models (transient)	Iterative-spatial representation over time of hydrogeology based on flow equations	Allows greater representation of real world, takes account of storage effects	Increased development time, complexity, data, misuse
Numerical quality models (transport)	Model of groundwater quality, temperature and age based on transport equations	Allows greater representation of real world, takes account of storage effects	Increased development time, complexity, data, misuse

4.5 Description of Individual Methods

Ten methods to assess groundwater ecological flows and water levels are described briefly, with advantages and disadvantages summarised. Each methodology is outlined and 'decision pathway to setting ecological flows' is provided. A recommendation on the circumstances in which the method should be applied is also made, cross-referencing to Table 4.2.

Decisions on ecological flows and water levels in groundwater systems require increasing scientific knowledge as the significance of resource values increases and as the user pressure on the groundwater system grows. Therefore, the 'decision pathway' commonly includes the application of more than one method so that scientific knowledge of a groundwater system builds in response to increasing knowledge demands.

4.5.1 Conceptual Model/Simple Water Balance

Framework for use (Table 4.2):

Hydrol. alteration	Values L	M	H
L	✓	✓	
M	✓		
H			

a. Description

A conceptual model, inclusive of a simple water balance, is a basic representation of the components of the aquifer system (Anderson and Woessner 1992). The model includes all readily available information, and idealised concepts, referenced from literature such as: system boundaries, the hydrogeology and physical nature of the aquifer, the components of groundwater recharge, components of groundwater flow and components of groundwater discharge.

This method is an initial approach to assessing ecological flows and water levels in groundwater and provides a basis for further assessment. The 'decision pathway to setting ecological flows' includes a conceptual model at an early stage of all methods because conceptual models are the first step in groundwater assessment. Advantages of conceptual models include simplicity of data needs and an ability to provide an overall hydrological framework around the setting of ecological flows. A disadvantage of this method is that information availability is poor for many New Zealand aquifer systems, resulting in uncertainty in application as well as in outcomes.

b. Methodology

The most basic information can provide an assessment of groundwater system behaviour and it is likely that a range of information may be available to provide for conceptual models and water balances. The physical parameters and boundaries of the aquifer system (grouped as a single hydrogeological unit or as separate aquifers for groundwater management) should be identified.

The conceptual model includes a characterisation of a groundwater system including the following components:

- geology (eg, type of aquifer, basement)
- aquifer type (eg, unconfined, confined)
- aquifer extents (lateral and vertical)
- likely recharge sources
- likely discharge locations (including streams, lakes, sea, and groundwater abstraction)
- groundwater level
- flow directions
- inter-aquifer groundwater transfer
- groundwater quality.

The simple water balance includes a characterisation of a groundwater system including the following components for estimating:

- likely recharge rates (eg, from rivers, rainfall and irrigation)
- likely flow rates with some simple approaches (eg, based on Darcy's law)
- likely discharge rates (eg, to rivers, lakes, sea and from groundwater; abstraction) and using some simple approaches (eg, stream flow gaugings)
- groundwater volumes
- inter-aquifer groundwater transfer
- errors in rates of: groundwater recharge, groundwater discharge and groundwater flow.

It is important that the conceptual model and simple water balance are simplified, or approximated, in the correct context based on system complexity and scale. The water balance must provide for conservation of flow. Typically the water balance will represent average conditions of recharge, flow and discharge. However, changes in groundwater storage may also be identified in terms of time-variant water balances.

c. Decision pathway to setting ecological flows and water levels

1) Identify the boundaries of the groundwater system including top, bottom and lateral boundaries.
2) Apply the conceptual model/simple water balance method.
3) Identify sources of groundwater recharge and estimate rates of groundwater recharge.
4) Identify locations of groundwater discharge and rates of groundwater discharge.
5) Identify rates of groundwater flow.

6) Make decisions on ecological groundwater flows and levels that:
   – are suitably conservative
   – represent average conditions of recharge, discharge and flow in the aquifer
   – may represent time-varying conditions of recharge, discharge and flow in the aquifer
   – consider errors in rates of: groundwater recharge, groundwater discharge and groundwater flow
   – maintain ecological flows in surface water potentially linked to groundwater (see rivers, lakes and wetlands)
   – maintain inter-aquifer groundwater flows
   – include limits on groundwater allocation.

Recommendation:
A conceptual model/ simple water balance should always be applied as an initial method to assess groundwater resources. The approach provides an initial assessment to establish an ecological flow, or groundwater level.

4.5.2 Historical Levels

Framework for use (Table 4.2):

Hydrol. alteration	Values L	M	H
L	✓	✓	
M	✓		
H			

a. Description

Historical groundwater levels and groundwater level variations are examined in this method. Groundwater level data may be grouped to form piezometric contour information to provide an understanding of spatial variability of groundwater level and an understanding of the variability of groundwater level with time. The data may be plotted in a time series to assess level trends over time.

The typical applications of assessing historical levels are to:

- maintain groundwater outflows that sustain surface water
- maintain groundwater ecology (flora and fauna)
- maintain groundwater quantity
- assistance with maintaining groundwater quality
- and to prevent saltwater intrusion via direct saline ingression or up-welling.

Critical factors to be assessed from this method include:

- groundwater levels
- groundwater level variation in time
- groundwater level variation within aquifers and aquifer systems.

The advantages of the method include simplicity of approach as levels are directly related to aquifer recharge and discharge. Disadvantages of the method include a commonly unknown quality of historical data: groundwater levels alone may not be sufficient to determine allocation for the aquifer system.

The method can contribute to:

- maintaining outflows that sustain surface water by ensuring that the local and regional groundwater level is at a sufficient height above the level of the receiving surface waters in connection with the aquifer for a sufficient (to be determined) period of time
- maintaining groundwater ecology (flora and fauna) by ensuring that the local and regional seasonal groundwater level variation remains within a suitable range and that long-term groundwater levels are within a suitable range to provide for ecological requirements
- preventing saltwater intrusion by ensuring for a suitable duration that the groundwater level is of sufficient height above levels in coastal surface waters
- maintaining groundwater quantity, and assist with maintaining groundwater quality, by ensuring that groundwater levels remain within a suitable range to ensure that ecological flows and water levels in aquifers are maintained; and that relatively poor-quality groundwater, or poor-quality surface water, is not drawn into an aquifer.

b. Methodology

- Collate historical groundwater level (ie, hydraulic head) data; assess the quality of this data and assess its data for analysis of groundwater levels and groundwater level variations on short-term, seasonal, medium-term and long-term time scales.
- Collate historical surface water level information and assess the suitability of this data for analysis of surface water levels and surface water level variations on short-term, seasonal, medium-term and long-term time scales.
- Assess the groundwater level and surface water level data together for suitability of identifying interaction between groundwater and surface water.
- Assess errors in groundwater level measurements.
- Assess errors in surface water level measurements.
- Establish time series plots of historical groundwater level measurements.
- Establish time series plots of historical surface water level measurements.
- Establish groundwater level maps including levels of relevant surface water features – identify groundwater flow directions and identify if possible the relationships between groundwater and surface water.

Ecological water levels in the aquifer system may be set by assessing groundwater level time series, or by visual inspection of a groundwater level map, to identify:

- short-term, or local, groundwater levels that relate to groundwater abstraction
- seasonal, or medium-term groundwater levels that relate to seasonal groundwater abstraction or seasonal groundwater recharge
- long-term, regional groundwater level trends that relate to long-term groundwater sustainability
- identification of surface waters that potentially receive groundwater or lose water to groundwater and that are possibly dependent on groundwater as a source of baseflow.

Confidence limits of groundwater level time assessments should be assessed by considering errors or gaps in groundwater level measurements.

c. Decision pathway to setting ecological flows and water levels

1) Identify the boundaries of the groundwater system including top, bottom and lateral boundaries.
2) Apply the conceptual model/simple water balance method
3) Apply the historical levels method
4) Identify sources of groundwater recharge and estimate rates of groundwater recharge
5) Identify locations of groundwater discharge and rates of groundwater discharge
6) Identify rates of groundwater flow from the simple water balance
7) Make decisions on ecological groundwater flow rate and ecological groundwater level that:
   - are suitably conservative
   - represent average conditions of recharge, discharge and flow in the aquifer
   - may represent time-varying conditions of recharge, discharge and flow in the aquifer
   - consider errors in rates of: groundwater recharge, groundwater discharge and groundwater flow
   - maintain relative levels of groundwater and surface water so that natural groundwater recharge, or natural groundwater discharge, continues
   - maintain relative levels of groundwater in aquifers so that natural inter-aquifer groundwater transfers continue
   - maintain ecological flows in surface water potentially linked to groundwater (see rivers, lakes and wetlands)
   - include limits on groundwater allocation.

Recommendation: Historical levels should always be assessed whenever there is suitable historical level information available to make useful interpretations of the response of groundwater level to the natural variability of recharge, the variability of groundwater use, the locations of groundwater recharge and the locations of groundwater discharge. The method is especially applicable to provide first estimates of groundwater recharge,

from changes in groundwater storage over time, and groundwater flow, from estimates of groundwater level gradients.

4.5.3 Expert Panel

Framework for use (Table 4.2):

Hydrol. alteration	Values L	M	H
L	✓	✓	
M	✓		
H			

a. Description

An independently appointed panel of experts can provide advice on ecological flows and levels in groundwater and on ecological flows in surface waters supported by groundwater discharge.

This method has the advantages that it is quick, cheap, has credibility (dependent on experts), and can help overcome mistrust if well managed. The disadvantages of this method include that the expert panel can be used as a political tool and that implied consensus of the expert panel can lead to poor environmental outcomes.

Typical applications of expert panels are for:

- maintaining outflows that sustain surface water
- maintenance of groundwater ecology (flora and fauna)
- controlling land subsidence and aquifer consolidation
- controlling saltwater intrusion
- maintaining groundwater quality.

The critical factors that need to be considered are:

- aquifer head/variation
- gradient
- point and non-point sources
- flow
- land use.

b. Methodology

An expert panel may be called to assess a groundwater system with respect to ecological flows and levels in groundwater, and on ecological flows in surface waters supported by groundwater discharge. This type of assessment should include all the aspects, and include all existing knowledge, of the groundwater system.

Assessments of this nature are generally on an agreed approach by more than one expert appointed to the panel. The panel reviews all existing information including: conceptual model, water balance and detailed models of geology, recharge, flow and chemistry.

The panel aims to address ecological flows and levels in groundwater and in groundwater that supports surface water flows by assessment of:

- factors included in the conceptual model/simple water balance method
- factors included in the historical levels method
- any other relevant information considered important by the Expert Panel such as groundwater flow models and groundwater quality models.

An expert panel may be called at any stage of an assessment of ecological flows and water levels. The above approach is particularly suitable initially to provide sufficient review of available hydrogeological data for the setting of ecological flows, or levels, in groundwater; and in setting ecological flows where groundwater discharge supports surface water flow. A team approach to initial assessments commonly provides good results, particularly with initial assessments, because the experience of a range of groundwater experts and surface water experts can efficiently identify all relevant technical requirements. The expert panel should also assess gaps in knowledge and consider future research needs; in this way the knowledge base will improve over time to meet future requirements for improved knowledge of a groundwater system.

The expert panel provides a quick and cheap method that builds consensus with stakeholders. However, its effectiveness is limited by the credibility of the experts and poor outcomes can result from the need for consensus.

c. Decision pathway to setting ecological flows and water levels

1) Identify the boundaries of the groundwater system including top, bottom and lateral boundaries.
2) Apply the expert panel method.
3) Identify sources of groundwater recharge and estimate rates of groundwater recharge.
4) Identify locations of groundwater discharge and rates of groundwater discharge.
5) Identify rates of groundwater flow from the simple water balance.
6) Make decisions on ecological groundwater flows and levels that:
    - are suitably conservative
    - represent average conditions of recharge, discharge and flow in the aquifer
    - may represent time-varying conditions of recharge, discharge and flow in the aquifer
    - consider errors in rates of: groundwater recharge, groundwater discharge and groundwater flow
    - maintain relative levels of groundwater and surface water so that natural groundwater recharge, or natural groundwater discharge, continue

- maintain relative levels of groundwater in aquifers so that natural inter-aquifer groundwater transfers continue
- maintain ecological flows in surface water potentially linked to groundwater (see rivers, lakes and wetlands)
- include limits on groundwater allocation.

Recommendation: An expert panel should be considered in the early phases of a groundwater assessment for review, initial assessment of ecological flows or groundwater levels, and identification of future research needs. An expert panel may be considered in the later phases of a groundwater assessment in a review capacity.

4.5.4 Detailed Water Balance

Framework for use (Table 4.2):

Hydrol. alteration	Values L	M	H
L			✓
M		✓	
H	✓		

Critical factors: Aquifer head-water volumes.

a. Description

This method is used for quantifying groundwater flows, including inflows and outflows and has the advantage that it may be applied to all situations. Disadvantages of the method include that assumptions about groundwater flows are commonly required: some components of the balance such as subsurface recharge and subsurface discharge are generally unknown or very uncertain. A detailed water balance approach is more complex than a simple water balance approach, in that: specific recharge investigations and outflow measurements, including the metering of groundwater abstraction, may be made; and water balances over time are calculated. Thus, the detailed water balance is likely to include aquifer geometry, hydrogeologic parameters, boundaries and stresses.

Specific requirements for the water balance include:

- recharge sources and quantification of spatial and temporal water input
- aquifer discharge zones and the relationship to adjoining streams and rivers
- the size of the groundwater resource including volume of groundwater storage
- the effects of adjacent boundaries (possibly recharge and discharge boundaries)
- system stresses including groundwater abstraction over time.

The detailed water balance refers to the conceptual model and includes estimates of recharge from rivers (White et al. 2001) rainfall, and irrigation (Thorpe 2001) and discharge to surface water (White et al. 2001).

b. Methodology

Technical approaches to estimating groundwater recharge include (but are not limited to) the following:

- rainfall recharge to groundwater may be estimated by infiltration analyses, soil water balance models, catchment runoff analyses, environmental isotopes and chemical tracers
- recharge to groundwater from surface water (eg, river, lake, sea) may be estimated by stream flow gaugings, Darcy's law or stream tube analysis supported with sufficient hydraulic data, environmental isotopes and chemical tracers
- recharge to groundwater from irrigation may be estimated by infiltration analysis, environmental isotopes and chemical tracers
- recharge to groundwater through the sub-surface may be estimated by Darcy's law or flow net analysis with sufficient hydraulic data, environmental isotopes and chemical tracers.

Technical approaches to estimating groundwater flow and groundwater storage include (but are not limited to) the following:

- groundwater flow may be estimated by using Darcy's law or flow net analysis coupled to estimates of hydraulic conductivity or transmissivity from aquifer pumping tests, environmental isotopes and chemical tracers
- groundwater storage volumes may be estimated by aquifer pumping tests or regional estimates based on recharge analysis or aquifer response to air pressure or sea level variation where sufficient groundwater level measurements are available.

Technical approaches to estimating groundwater discharge to surface waters include (but are not limited to) the following:

- groundwater discharge to rivers and streams water may be estimated by stream flow gaugings or using Darcy's law or flow net analysis supported with sufficient hydraulic data
- groundwater discharge to lakes and wetlands may be estimated by using Darcy's law or flow net analysis supported with sufficient hydraulic data.

Detailed information, such as stream gaugings, may indicate whether a stream is gaining water from, or losing water to, an aquifer. The physical parameters and boundaries of the aquifer system (grouped as a single hydrogeological unit, or individual aquifer, for groundwater management) should be specifically identified, including groundwater inflow and natural recharge from rainfall or stream flow, artificial recharge, vertical leakage and excess irrigation water. Groundwater losses should also be identified from the system including evapotranspiration, vertical leakage and abstraction, groundwater discharge and change in storage.

The water balance obtained must provide for conservation of flow/volume, be calibrated against known measurements and for modelling purposes, must have convergence. Changes in water storage may also be identified in terms of transient approaches to the conceptual model/water balance.

Application of the detailed water balance is at a regional scale and it includes:

- assessment of effects of cumulative groundwater abstraction on groundwater discharge and flow in spring-fed streams over a relatively coarse time scale so that ecological flows in spring-fed streams remain at acceptable levels
- assessment of cumulative groundwater use on groundwater levels and groundwater flows over time so that ecological flows and water levels within groundwater systems are maintained.

c. Decision pathway to setting ecological flows and water levels

1) Identify the boundaries of the groundwater system including top, bottom and lateral boundaries
2) Apply the conceptual model/simple water balance method
3) Apply the historical levels method
4) Apply the detailed water balance method
5) Identify sources of groundwater recharge and estimate rates of groundwater recharge
6) Identify locations of groundwater discharge and rates of groundwater discharge
7) Identify rates of groundwater flow from the simple water balance
8) Make decisions on ecological groundwater flow rate and groundwater levels that:
   - are suitably conservative
   - consider errors in rates of: groundwater recharge, groundwater discharge and groundwater flow
   - maintain relative levels of groundwater and surface water so that natural groundwater recharge, or natural groundwater discharge, continues
   - maintain relative levels of groundwater in aquifers so that natural inter-aquifer groundwater transfers continue
   - maintain ecological flows in surface water potentially linked to groundwater (see rivers, lakes and wetlands)
   - consider groundwater allocation options by location and in time
   - consider potential effects of groundwater allocation options on ecological flows and water levels
   - include limits on groundwater allocation.

Recommendation: A detailed water balance should be developed to assess ecological flows, or water levels. Groundwater recharge, groundwater flow, groundwater storage and groundwater discharge may be estimated (and uncertainty of the estimates quantified) provided observed data are of sufficient quantity and quality.

4.5.5 Time Series Analysis

Framework for use (Table 4.2):

Hydrol. alteration	Values L	M	H
L			✓
M		✓	
H	✓		

a. Description

This method includes statistical analyses of: groundwater levels, groundwater system inputs, groundwater system outputs and statistical analysis of groundwater quality over time. The method assists with: maintenance of groundwater flow, maintenance of groundwater outflows that sustain surface water, maintenance of groundwater ecology (flora and fauna), controlling saltwater intrusion, maintenance of groundwater quality.

An advantage of the method is that it is relatively easy to apply and scenarios of environmental variability and groundwater use can be tested. Disadvantages include:

- dependence on observed data – poor quality results can come from poor data
- the duration of groundwater level data may be short and not sufficient for time series analysis
- groundwater level measurements may be too infrequent for time series analysis
- observation wells be in a poor location for identifying the drivers of groundwater level variation
- groundwater system inputs and outputs may be poorly known and adequate data on groundwater system inputs and outputs may not be available.

The disadvantages of the method for assessing groundwater quality data include groundwater quality data that are not collected with standard sampling techniques and poor laboratory analysis as indicated by ion balances.

Time series analysis contributes to the applications through identifying the effects of groundwater inputs on groundwater level, or groundwater discharge. For example simple correlation of base flow discharge in streams with rainfall recharge may be useful to assessment of ecological flows/water levels where rainfall recharge is declining in the long term.

b. Methodology

Time series analysis aims to relate responses of groundwater systems to stressors on groundwater systems. Groundwater system responses include: groundwater level (ie, hydraulic head), groundwater discharge (eg, base flow stream discharge) and groundwater quality. Groundwater system stressors include groundwater recharge from rainfall, or from rivers, and groundwater use.

Many approaches are available, including:

- simple correlation
- Fourier time-series analysis (White 1994)
- Principal Component Analysis (Cameron and White 2004)
- river recharge and rainfall recharge separation (White and Brown 1995)
- neural networks (White et al. 2003)
- the 'eigenvalue' model (Bidwell and Morgan 2001).

Typically, the analysis process follows:

- potential groundwater system inputs (eg, recharge from rainfall and from rivers) or outputs (eg, groundwater discharge to rivers and lakes) are identified in the conceptual model. The characteristics of inputs and outputs are identified, or estimated, over time. For example rainfall recharge is a seasonal input to groundwater on the east coast of New Zealand because rainfall recharge during summer seasons is typically lower than rainfall recharge during winter seasons (White et al. 2003)
- groundwater levels, and groundwater quality are plotted and commonly interpolated into a constant time base for ease of analysis – the time base chosen is relevant to identification of groundwater level response over short, seasonal, medium and long-term time scales
- groundwater level variations on short, seasonal, medium and long-term time scales are identified. The causes of these level variations are then identified. For example a groundwater level variation over a week, or less, duration may be due to pumping groundwater from a neighbouring well
- relationships between groundwater system inputs (eg, magnitude and period) and groundwater level response (eg, magnitude, period and time lags of response) are expressed as equations
- a relation between groundwater level response and a groundwater system input may then be explored.

Statistical analysis of groundwater quality over time is used to assess groundwater quality changes (eg, in response to land use change). Typically the changes of relevant individual ionic species are assessed (eg, nitrate-nitrogen for land use and chloride for salt water intrusion) in time-series plots; Piper diagrams (Rosen 2001) may be used to assess changes of suites of ions over time.

c. Decision pathway to setting ecological flows and water levels

1) Identify the boundaries of the groundwater system including top, bottom and lateral boundaries.
2) Apply the conceptual model/simple water balance method.
3) Apply the historical levels method.
4) Apply the time series analysis method.

5) Identify sources of groundwater recharge and estimate rates of groundwater recharge.
6) Identify locations of groundwater discharge and rates of groundwater discharge.
7) Identify rates of groundwater flow from the simple water balance.
8) Make decisions on ecological groundwater flow rate and groundwater levels that:
   – are suitably conservative
   – consider errors in rates of: groundwater recharge, groundwater discharge and groundwater flow
   – maintain relative levels of groundwater and surface water so that natural groundwater recharge, or natural groundwater discharge, continues
   – maintain relative levels of groundwater in aquifers so that natural inter-aquifer groundwater transfers continue
   – maintain ecological flows in surface water potentially linked to groundwater (see rivers, lakes and wetlands)
   – consider groundwater allocation options by location and in time
   – consider potential effects of groundwater allocation options on ecological flows and water levels
   – include limits on groundwater allocation.

Recommendation: Time series analysis should be used to assess ecological flows, or groundwater levels, with a statistical analyses including: groundwater levels, groundwater system inputs, groundwater system outputs and groundwater quality over time. Time series analysis also provides key information on groundwater resource sustainability, including allocation decisions, over time.

4.5.6 Analytical Models

Framework for use (Table 4.2):

Hydrol. alteration	Values L	Values M	Values H
L			✓
M		✓	
H	✓		

a. Description

This method typically uses spreadsheet-based models that use groundwater flow and groundwater transport equations. The method assists with:

- maintaining groundwater flow and maintaining groundwater outflows that sustain surface water
- maintenance of groundwater ecology (flora and fauna)
- controlling land subsidence and aquifer consolidation

- controlling saltwater intrusion, and maintaining groundwater quality.

Advantages of this method include: ease of use, moderate skill level, moderate data requirements. Disadvantages of the method include: a common requirement for simple assumptions; cumulative effects of groundwater use may be ignored.

Analytical models assist the applications by assessing:

- groundwater pumping to ensure that ecological groundwater flows or levels are maintained, eg, to prevent saltwater intrusion
- groundwater pumping to ensure that ecological surface water flows are maintained;
- groundwater pumping and aquifer consolidation
- land use so that groundwater quality and surface water quality are maintained where surface water is linked to groundwater.

b. Methodology

Analytical models are solutions to the equations governing groundwater flow and solute transport in groundwater. These solutions are equations that are often implemented in spreadsheets.

Groundwater flow is assessed by analytical models that solve for groundwater flow, solute transport and boundary conditions. Boundary conditions represent the type of problem, including:

- pumping tests where groundwater flow to a well is assessed (Kruseman and de Ridder 1991) to estimate properties of the formation relating to water flow in porous materials and properties of the well
- groundwater–surface water interaction where surface water flow may be reduced when groundwater is pumped (Hunt 1999; Jenkins 1977; Pulido-Velazquez et al. 2005)
- solute transport to estimate chemical dilution in groundwater (Environment Canterbury 2007)
- ground consolidation associated with groundwater pumping at the local scale or regional scale.

Analytical models use parameters related to groundwater flow, solute transport and boundary conditions, including:

- aquifer type (unconfined or confined)
- well diameter
- pumping rate
- aquifer porosity
- aquifer hydraulic conductivity
- aquifer thickness,
- aquifer storability
- distance between surface water and pumping well

- dispersion coefficients and land use coefficients.

c. Decision pathway to setting ecological flows and water levels

1) Identify the boundaries of the groundwater system including top, bottom and lateral boundaries.
2) Apply the conceptual model/simple water balance method.
3) Apply the historical levels method.
4) Apply the analytical method.
5) Identify sources of groundwater recharge and estimate rates of groundwater recharge.
6) Identify locations of groundwater discharge and rates of groundwater discharge.
7) Identify rates of groundwater flow from the simple water balance.
8) Make decisions on ecological groundwater flow rate and groundwater levels that:
   - are suitably conservative
   - consider errors in rates of: groundwater recharge, groundwater discharge and groundwater flow
   - maintain relative levels of groundwater and surface water so that natural groundwater recharge, or natural groundwater discharge, continues
   - maintain relative levels of groundwater in aquifers so that natural inter-aquifer groundwater transfers continue
   - maintain ecological flows in surface water potentially linked to groundwater (see rivers, lakes and wetlands)
   - consider groundwater allocation options by location and in time
   - consider potential effects of groundwater allocation options on ecological flows and water levels
   - include limits on groundwater allocation.

Recommendation: Analytical models may be used to make decisions on ecological groundwater flows and groundwater levels. Analytical models may be used to assess pumping tests, groundwater–surface water interaction, solute transport, etc. However, uncertainty in model calculations may be large.

4.5.7 Numerical Quantity Models – Steady State

Framework for use (Table 4.2):

Hydrol. alteration	Values L	M	H
L			✓
M		✓	✓
H	✓	✓	✓

a. Description

This method represents aquifers and groundwater flow with a computer model based on groundwater flow equations and boundary conditions where model properties are constant in time. The method is relevant to:

- maintaining outflows that sustain surface water
- maintenance of groundwater ecology (flora and fauna)
- controlling saltwater intrusion.

Numerical models of groundwater quality are built on estimates of groundwater flows and boundary conditions provided by numerical models of groundwater quantity.

Advantages of the method include:

- numerical models allow a complex representation of the real world and allow an assessment of many types of groundwater issues
- numerical models allow representation of groundwater flow in multi-aquifer groundwater basins.

Disadvantages of the method include:

- numerical models can be time-consuming to develop and they can be complex
- numerical models are intensive users of data yet data may be of poor, or unknown, quality
- model properties are commonly assumed because the steady-state data are often collected at a scale that is coarser than the model grid
- data collected at the local scale (eg, results from pump tests) may not be representative of model properties at the regional scale
- model boundary conditions may be poorly defined
- estimates of groundwater system behaviour, in a model application, may be poor even where model calculations agree well with steady-state data.

b. Methodology

Numerical models are solutions to the equations governing groundwater flow (Anderson and Woessner 1992) that aim to represent the variability of natural systems in two dimensions or three dimensions. Steady state numerical models aim to represent groundwater flow in average conditions, eg, average annual conditions of groundwater recharge and groundwater discharge. Numerical models are often developed with a graphical user interface and often applied in spreadsheets.

The method of numerical modelling may follow Anderson and Woessner (1992) to include:

1) development of a conceptual model of the system
2) development of model datasets representing components of groundwater hydrology including:
   - geology
   - aquifer type (eg, unconfined or confined) and aquifer properties
   - steady-state boundary conditions such as groundwater recharge (eg, from rainfall, irrigation and rivers)
   - steady-state boundary conditions such as groundwater discharge (eg, to springs, streams, lakes and sea) and groundwater use
3) calibration of the model that aims to provide a good representation of steady-state data
4) calibration sensitivity analysis that aims to assess uncertainty in the calibrated model
5) model application.

Steady-state numerical models may contribute to the applications, at local and regional scales, by assessing the:

- effects of pumping on groundwater discharge and flow in spring-fed streams so that ecological flow in spring-fed streams remains at acceptable levels
- assessment of cumulative pumping on groundwater levels and groundwater flows so that ecological flows and water levels within groundwater are maintained
- effects of groundwater pumping on the potential for salt water intrusion so that groundwater levels are maintained above sea level.

c. Decision pathway to setting ecological flows and water levels

1) Apply the numerical quantity models – steady-state method.
2) Complete an uncertainty analysis that assesses the effect of variability in model properties and model stressors (recharge and groundwater pumpage) on model estimates of groundwater level, groundwater discharge and groundwater flow.
3) Make decisions on ecological groundwater flow rate and groundwater levels that:
   - are suitably conservative
   - consider errors in rates of: groundwater recharge, groundwater discharge and groundwater flow

– maintain relative levels of groundwater and surface water so that natural groundwater recharge, or natural groundwater discharge, continues
– maintain relative levels of groundwater in aquifers so that natural inter-aquifer groundwater transfers continue
– maintain ecological flows in surface water potentially linked to groundwater (see rivers, lakes and wetlands)
– consider uncertainty in model calculations
– consider groundwater allocation options by location
– consider potential effects of groundwater allocation options on ecological flows and water levels
– include limits on groundwater allocation.

Recommendation: Steady-state numerical models of groundwater quantity should be applied to the setting of ecological groundwater flows and groundwater levels, where knowledge of a groundwater system is reasonably advanced. Ecological groundwater flows and groundwater levels should be assessed from a calibrated model with an analysis of uncertainty. Steady-state numerical models of groundwater quantity should be built and tested before development of transient models of groundwater quantity.

4.5.8 Numerical Quantity Models – Transient

Framework for use (Table 4.2):

Hydrol. alteration	Values L	M	H
L			✓
M			✓
H	✓	✓	✓

a. Description

This method represents aquifers and groundwater flow with a computer model based on groundwater flow equations and boundary conditions where model properties are variable in time. Numerical models of groundwater quality are built on estimates of groundwater flows and boundary conditions provided by numerical models of groundwater quantity.

Advantages of the method include:

- complex representations of real world are allowed
- many types of groundwater issues may be assessed
- transient numerical models allow representation of groundwater flow in multi-aquifer groundwater basins
- consideration of groundwater storage effects such as the time-dependent response of groundwater discharge to surface water such as may be observed in short and long term depletion of baseflow.

Disadvantages of the method include:

- models are time-consuming to develop and are commonly complex
- models are intensive users of data yet data may be of poor, or unknown, quality
- model properties are commonly assumed because the model data are often collected at a scale that is coarser than the model grid
- data collected at the local scale (eg, results from pump tests) may not be representative of model properties at the regional scale
- transient data is often smoothed or interpolated from observations
- model boundary conditions may be poorly defined
- estimates of groundwater system behaviour, in a model application, maybe poor even where model calculations agree well with transient data.

The method is relevant to:

- maintaining outflows that sustain surface water
- maintenance of groundwater ecology (flora and fauna)
- controlling saltwater intrusion.

b. Methodology

Numerical models are solutions to the equations governing groundwater flow (Anderson and Woessner 1992) that aim to represent the variability of natural systems in two dimensions or three dimensions over time. Transient numerical models aim to represent groundwater flow in typical conditions, eg, time-variable groundwater recharge and groundwater discharge. Numerical models are often developed with a graphical user interface.

The method of numerical modelling may follow Anderson and Woessner (1992) to include:

1) development of a conceptual model of the system
2) development of model datasets representing components of groundwater hydrology including:
   - geology
   - aquifer type (eg, unconfined or confined) and aquifer properties
   - transient boundary conditions such as groundwater recharge (eg, from rainfall, irrigation and rivers)
   - transient boundary conditions such as groundwater discharge (eg, to springs, streams, lakes and sea) and groundwater use
3) calibration of the model that aims to provide a good representation of transient data
4) calibration sensitivity analysis that aims to assess uncertainty in the calibrated model
5) model application.

Transient numerical models may contribute to the applications, at local and regional scales, by assessing the:

- effects of pumping on groundwater discharge and flow in spring-fed streams over time so that ecological time-varying flow in spring-fed streams remains at acceptable levels
- assessment of cumulative pumping on groundwater levels and groundwater flows over time so that time-varying ecological flows and levels within groundwater are maintained
- effects of groundwater pumping on the potential for salt water intrusion over time so that groundwater levels are maintained above sea level.

Numerical models of groundwater quality are built on estimates of groundwater flows and boundary conditions provided by numerical models of groundwater quantity.

c. Decision pathway to setting ecological flows and water levels

1) Apply the numerical quantity models – steady-state method.
2) Apply the numerical quantity models – transient method.
3) Complete an uncertainty analysis that assesses the effect of variability in model properties and model stressors (recharge and groundwater pumpage) on model estimates of groundwater level, groundwater discharge and groundwater flow.
4) Make decisions on ecological groundwater flow rate and groundwater levels that:
   - are suitably conservative
   - consider errors in rates of: groundwater recharge, groundwater discharge and groundwater flow
   - maintain relative levels of groundwater and surface water so that natural groundwater recharge, or natural groundwater discharge, continues
   - maintain relative levels of groundwater in aquifers so that natural inter-aquifer groundwater transfers continue
   - maintain ecological flows in surface water potentially linked to groundwater (see rivers, lakes and wetlands)
   - consider uncertainty in model calculations
   - consider groundwater allocation options by location and in time
   - consider potential effects of groundwater allocation options on ecological flows and water levels
   - include limits on groundwater allocation.

Recommendations: Transient numerical models should be used to set ecological flows, and groundwater levels, where knowledge of a groundwater system is advanced. Ecological groundwater flows and groundwater levels should be assessed from a calibrated model with an analysis of uncertainty.

4.5.9 Numerical Quality Models – Transport

Framework for use (Table 4.2):

Hydrol. alteration	Values L	Values M	Values H
L			✓
M			✓
H	✓	✓	✓

a. Description

Numerical quality models, more commonly known as 'contaminant transport models' are used to solve the partial differential advection dispersion equations for an entire flow field of interest. Algebraic equations for each model sub-area (or cell) are solved numerically through an iterative process for various transport options such as advection, dispersion, and chemical reactions including bacterial decay. This is also true for density-dependent models as commonly applied to assessing coastal saltwater intrusion. Such numerical quality models are usually coupled to flow models.

This method may model groundwater quality, groundwater temperature or groundwater age, based on groundwater transport equations and boundary conditions. Applications of the method include:

- controlling saltwater intrusion
- maintaining groundwater quality and
- maintaining surface water quality.

Advantages of transient numerical transport models include, that they allow:

- a complex representation of the real world and allow an assessment of many types of groundwater issues
- representation of multi-species transport in multi-aquifer groundwater basins and may also account for density-dependent groundwater flow as occurs with saline intrusion.

Disadvantages of the method include:

- these models are time-consuming to develop and they can be complex
- they are intensive users of data yet data may be of poor, or unknown quality
- obtaining a good calibration for the model is often difficult
- model properties are commonly assumed because the model data are often collected at a scale that is coarser than the model grid
- data collected at the local scale (eg, dispersion properties) may not be representative of model properties at the regional scale
- model transient data is often smoothed or interpolated from observations

- model boundary conditions may be poorly defined.

The advantage of using numerical contaminant transport models over their analytical counterparts is that they allow a greater variability of physical flow and transport parameters (which dominate transport movement) to be represented. It should be noted that specific 'site' concentrations might not be accurately predicted by these models due to errors in measurement and spatial variability in transport parameters. However, geostatistical tools may be used to interpret model results and reduce predictive uncertainty.

Examples of such applications of transport models are:

- management and remediation of a contaminated site where an existing or pre-existing contaminant source is identified and sufficient hydro-geological and geochemical information exists to predict local and/or regional impact of the specific contaminant plume. This type of application may require a management limit on source flow, concentration and/or groundwater level
- predicting the coastal saline interface for an unconfined or confined aquifer to determine minimum pumping levels or ecological levels in aquifers near the coast.

The transport modelling approach may be applied as a comprehensive assessment to appreciating groundwater responses of water quality to recharge or discharge, or introduced contaminant levels. This approach may allow an appropriate ecological level of groundwater flow. Land use effects on groundwater quality (Bidwell 2005), and on the quality of groundwater discharge to surface waters (White and Daughney 2002), may be assessed with spreadsheet models based on groundwater flow models and nutrient transport in groundwater.

b. Methodology

The process of numerical transport modelling may follow Anderson and Woessner (1992) to include:

1) development of a conceptual model of the system
2) development of model datasets representing components of groundwater transport and quality including (but not limited to):
   - all components of physical flow for steady state or transient numerical; quantity models as specified above
   - aquifer porosity distribution
   - aquifer dispersion properties
   - source recharge and discharge concentrations
   - chemical reactions including microbial decay within the aquifer.
3) calibration of the model that aims to provide a good representation of transient monitoring data
4) calibration sensitivity analysis that aims to assess uncertainty in the calibrated model
5) model application

6) making decisions on ecological groundwater flow rates and groundwater levels that:
   - are suitably conservative
   - consider errors in rates of: groundwater recharge, groundwater discharge and groundwater flow
   - maintain relative levels of groundwater and surface water so that natural groundwater recharge, or natural groundwater discharge, continues
   - maintain relative levels of groundwater in aquifers so that natural inter-aquifer groundwater transfers continue
   - maintain ecological flows in surface water potentially linked to groundwater (see rivers, lakes and wetlands)
   - consider uncertainty in model calculations
   - consider groundwater allocation options by location and in time
   - consider potential effects of groundwater allocation options on ecological flows and water levels
   - maintain, or improve, groundwater quality
   - include limits on groundwater allocation.

c. Decision pathway to setting ecological flows and water levels

1) Apply the numerical quantity models – steady-state method if the transport model is steady state.
2) Apply the numerical quantity models – transient method if the transport model is transient.
3) Complete an uncertainty analysis that assesses the effect of variability in model properties and model stressors (recharge and groundwater pumpage) on model estimates of groundwater level, groundwater discharge and groundwater flow.
4) Make decisions on ecological groundwater flow rate and groundwater level that:
   - are suitably conservative
   - consider errors in rates of: groundwater recharge, groundwater discharge and groundwater flow
   - maintain relative levels of groundwater and surface water so that natural groundwater recharge, or natural groundwater discharge, continues
   - maintain relative levels of groundwater in aquifers so that natural inter-aquifer groundwater transfers continue
   - maintain ecological flows in surface water potentially linked to groundwater (see rivers, lakes and wetlands)
   - maintain groundwater discharge across the coastal boundary to prevent salt water intrusion
   - consider uncertainty in model calculations
   - consider groundwater allocation options by location, if the transport model is steady state, and in both location and time if the transport model is transient

– consider potential effects of groundwater allocation options on ecological flows and water levels
– include limits on groundwater allocation.

Recommendation: Numerical quality models – transport should be used to set ecological groundwater flow water levels, where knowledge of a groundwater system is advanced. Ecological groundwater flows and groundwater levels should be assessed from a calibrated model with an analysis of uncertainty.

4.5.10 Consolidation Models

Framework for use (Table 4.2):

Hydrol. alteration	Values L	M	H
L			
M			✓
H		✓	✓

a. Description

This method aims to estimate settlement of ground materials caused by groundwater depressurisation. The method is applicable to controlling land subsidence and controlling aquifer consolidation.

An advantage of the method is that it allows quantification of consolidation. Disadvantages of the method include: much input data is required by sophisticated models; sophisticated models require experienced modellers; field measurements may not represent the variability of ground over short distances.

Applications of consolidation models include assessment of settlement risk to services and structures from groundwater level variations over time within the parent aquifer. Management of groundwater levels may be required where consolidation poses sufficient risk to infrastructure or to aquifer. This application may require a management limit on groundwater level, and groundwater use, to protect aquifer integrity and maintain groundwater levels in an aquifer.

The consolidation modelling approach could be applied as part of a comprehensive assessment of responses to groundwater level variation over time.

b. Methodology

Consolidation models generally utilise output from analytical or numerical flow models to estimate groundwater level variability over time. Consolidation within overlying materials, or within an aquifer, is assessed from groundwater level and from the properties of materials within a target area.

Assessment of consolidation is mainly in the form of:

- broad assessments – initial risk analysis from the comparison of groundwater level variation in relation to local unconsolidated geology
- engineering calculation – conventional consolidation calculations (Bowles 1996) and to determine site-specific or regional problems. Output in the form of consolidation settlement at a specific site or geo-statistical contoured distribution of consolidation settlement over a specified area
- application of 2D and 3D approaches with the use of finite element or finite difference iterative numerical modelling techniques. This type of modelling environment is generally limited to structural design assessments.

Broad assessments may be included in conceptual modelling whereby a first estimate of risk is made. Engineering calculation is defined as an analytical consolidation model. 2D and 3D approaches using finite element or finite difference numerical modelling techniques are likely to require conceptual and analytical approaches as a prerequisite.

c. Decision pathway to setting ecological flows and water levels

1) Development of a conceptual model of the system.
2) Development of model datasets representing components of groundwater hydrology and sediment properties including:
   - geology
   - sediment compaction factors (Lambe & Whitman 1969)
   - aquifer type (eg, unconfined or confined) and aquifer properties
   - boundary conditions such as groundwater recharge (eg, from rainfall, irrigation and rivers)
   - boundary conditions such as groundwater discharge (eg, to springs, streams, lakes and sea) and groundwater pumpage.
3) Calibration of the model that aims to provide a good representation of observed data.
4) Calibration sensitivity analysis that aims to assess uncertainty in the calibrated model.
5) Model application.
6) Make decisions on ecological groundwater flow rate and groundwater level that:
   - are suitably conservative
   - consider errors in rates of: groundwater recharge, groundwater discharge, groundwater flow and ground properties
   - maintain relative levels of groundwater and surface water so that natural groundwater recharge, or natural groundwater discharge, continues
   - maintain relative levels of groundwater in aquifers so that natural inter-aquifer groundwater transfers continue
   - maintain ecological flows in surface water potentially linked to groundwater (see rivers, lakes and wetlands)
   - maintain groundwater discharge across the coastal boundary to prevent salt water intrusion

- constrain sediment compaction so that the risk of infrastructural damage is within acceptable limits
- include limits on groundwater allocation.

Recommendation: Consolidation models should be used to set ecological groundwater flow and groundwater levels, where knowledge of a groundwater system is advanced. Ecological groundwater flows and levels should be assessed from a calibrated model with an analysis of uncertainty.

5. References of Main Text and Appendices

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- Appendix 1

Relationship between Total Allocation and Ecological Flow Requirements in Rivers

Relationship between Total Allocation and Ecological Flow Requirements in Rivers

Ideally, the effects of water abstraction or any other flow manipulation on the natural flow regime would be known in order to determine specific requirements of the ecological flow regime. For example, there might be a requirement for flushing flows below impoundments; or where the level of allocation results in extended periods of low flow, the minimum flow might be set higher or flow-sharing options considered. Unfortunately, ecological flow requirements must often be determined without knowing potential out-of-stream uses.

The most common situation is probably where a relatively small amount of water is taken from a river, usually for irrigation and sometimes for town supply. This might amount to 10–20% of summer low flows, but not be a significant proportion of high flows and not a sufficiently large total allocation to reduce the stream flow to a minimum for extended periods of time. Summer is the critical period for this type of abstraction because this is when maximum abstraction usually occurs and when low flows and high water temperatures may limit biological communities.

As the amount of abstracted water increases, so does the potential to reduce river flows to a minimum for extended periods. This is not necessarily deleterious as is often assumed. If instream conditions at the minimum flow are adequate (ie, provide optimal habitat quality or habitat levels that occur with annual natural low flows), then biota should not be detrimentally affected, provided the frequency of higher flushing and channel maintenance flows remains unchanged. However, if instream conditions at minimum flow provide less than optimal habitat quality (eg, average habitat suitability index), an increase in the duration of low flows increases the risk of detrimental effects. Few studies have examined the effects of extended duration of low flow. Jowett et al. (2005) showed that in the Waipara River, where habitat is limited at low flow, the detrimental effect on fish numbers increased with the magnitude and duration of low flow. When summer flows were less than the mean annual low flow for about 30% of the time, there was a substantial decline in abundance of three of the four common native fish species in the river. When summer flows were less than mean annual low flow (MALF) for about 10% of the time, there was little change in native fish abundance. The effect was more severe on fast-water species (torrentfish and bluegill bullies) than species that prefer lower-velocity water (upland bullies and Canterbury galaxias).

If instream conditions at low flows are less than optimal, then increasing the duration of low flows through increasing total allocation increases the risk of detrimental effects. Obviously a reduction in the length of time that habitat is sub-optimal will reduce the detrimental effects. This can be done by increasing the minimum flow requirement (Figure A1.1) or by increasing the frequency of higher flows by a flow-sharing arrangement, whereby the amount of water available for abstraction at any particular time is some proportion of the natural flow less the minimum flow requirement. Either method reduces the total volume of water available for abstraction and the reliability of supply.

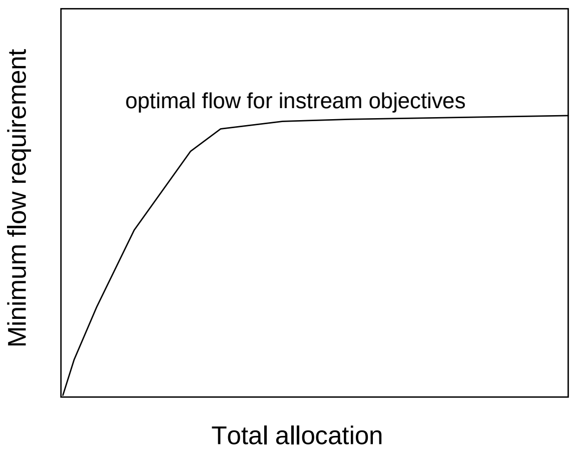

Figure A1.1: Conceptual relationship between minimum flow requirement and total allocation

In practice, Regional Councils have imposed limits on the total allocation that do not significantly extend the duration of low flows and guarantee a certain reliability of supply to consent holders. This raises the question of what level of abstraction will significantly alter the duration of low flows, and in what type of river will the duration of low flow have a significant detrimental effect on biota? A river risks deleterious effect from flow reduction if habitat quality at natural low flows is less than optimal. Instream habitat analyses show clearly that small streams have less than optimal habitat quality for salmonids and many native fish species, but that flows in larger rivers can be reduced to create optimal habitat. In addition, a 'small' stream for salmonids is larger than a 'small' stream for native fish, so allocation levels, as well as flow requirements, will depend on the species present.

Allocation Limits

Allocation limits have been used to manage water resources and a common approach is to limit total allocation to a proportion of a flow statistic such as the mean annual low flow. If the total allocation is low, the degree of hydrological alteration and thus ecological effect will be small. For example, if the total allocation is less than 10% of the MALF, abstractors will have high reliability of supply and there may be no need for any restriction such as a minimum flow requirement. This is because 10% of MALF is barely measurable with good flow measuring techniques and is therefore unlikely to have any biologically detectable effect.

The Motueka Conservation Order, 2004, limits abstraction to 12% of the instantaneous flow and presumably assumes that this will have negligible ecological effects. A 12% flow difference is just detectable by available flow measuring methods. This method of allocation guarantees that there is some water available for abstraction, even at lowest flows.

Another method of defining allocation limits is based on defining the acceptable level of risk to the environment and reliability of supply to the resource user. This is based on frequency and duration analyses of the hydrological record. For example, if the target reliability of supply is 95% and the frequency of the minimum flow is 1%, then the amount of water available for allocation is the difference between the flow that is exceeded for 94%

of the time and the minimum flow. When the total allocation is being fully used, the frequency of occurrence of the minimum flow increases to the frequency of occurrence of the minimum flow plus allocation (ie, 6% of the time in the example). In terms of days, the frequency of the minimum flow increases from about 4 days per year to 22 days per year.

The effect of allocation extending the duration of low flows should be considered in terms of biological significance. Are there likely to be significant biological effects with the change in duration? In considering this, the quality of the habitat at low flow should be taken into consideration as described in the preceding section. If habitat at low flow is sub-optimal and limiting biota, extending the duration of low flows is likely to increase the detrimental effect on biota. Increasing the minimum flow can mitigate the detrimental effect; this calculation is described by Jowett and Hayes (2004). If habitat at the minimum flow is optimal or higher (Figure A1.1), then the biological effects of abstraction are likely to be minor or even beneficial; and there is no need to limit allocation from an ecological perspective until the volumes abstracted affect the magnitude and duration of minor freshes.

▪ Appendix 2

Effects of Flow Regime on Stream Ecology

Effects of Flow Regime on Stream Ecology

The driving force of a stream is the current. It is necessary for the respiration of many benthic invertebrates and reproduction of some fish species (Hynes 1970). Currents distribute nutrients and food down a river system – detritus for invertebrates and drifting insects for fish and birds – and aid species dispersal. Biologists and anglers who study rivers are well aware that aquatic species are likely to be found associated with specific habitats; many aquatic species are found in similar hydraulic conditions in a wide range of rivers. These have been termed habitat niches and include both physical and biotic characteristics of the environment (Odum 1971). Such concepts have been widely applied in both terrestrial and aquatic biological studies and the presence of suitable habitat for any species is a necessary condition for survival.

Aquatic life in streams and rivers has developed under a 'natural' flow regime. If the instream environment under natural flows is unsuitable for a particular species then that species will not be well established in a stream. Periodic disturbances, such as floods and droughts, affect stream biota. Floods can reduce trout stocks (Jowett and Richardson 1989), invertebrates (Quinn and Hickey 1990), and periphyton (Biggs et al. 1990). However, the effect of disturbance frequency differs between aquatic species. If disturbances are too frequent and severe, most biota will be unable to establish self-sustaining populations. Native fish and brown trout seem to be particularly well adapted to surviving large floods, even taking advantage of the situation to feed (Jowett and Richardson 1994). Aquatic insects are also relatively robust, colonising a stream within 4–6 weeks of a severe disturbance (Sagar 1983; Scrimgeour and Winterbourn 1989). Some stream insects recolonise streams relatively quickly: as drifting insects from upstream, from within the gravels, or from eggs laid by the terrestrial adult insects. The recolonisation rate of fish is slower than of stream insects. Floods particularly affect juvenile trout and adult rainbow trout, presumably because they do not utilise cover as well as adult brown trout, and juvenile trout – especially recently emerged fry – are weaker swimmers.

The biota present in a stream have survived series of disturbances and, presumably, will continue to survive provided that the frequency of these disturbances does not change appreciably. Some stream ecologists hypothesise that stream biota have adapted to the flow regime of particular streams or rivers, and in particular, they believe that biota have adapted to, or survive, the low flows that occur in the river every year or so. If the abundance of an aquatic species in a particular stream is limited by the naturally occurring low flows in that stream, then further reduction in flow would have a detrimental effect on that species, but if the species is not limited by low flows then reduction in flow will have little effect. Given that the life of most stream fish is between 3 and 15 years, fish will have survived droughts that occur about once every two years; the status quo, in terms of stream biology, is likely to be retained if the minimum flow does not fall below the average natural low flow: the mean annual minimum flow.

New Zealand River Flow Regimes

Rivers in New Zealand vary greatly, influenced by our geographic and climate features, including the maritime location and tectonically young/active landscape. However, it is possible to categorise flow regimes into broad groups based on local climate, topography, watershed geology and land cover (Snelder et al. 2005). Examples of the flow regimes from three main river types in the South Island are:

Mountain source of flow – having relatively high minimum flows, with very frequent (often weekly) high flows of 3 to 6 times the low flow, and large amounts of transported gravel. Low flows typically occur in winter and seldom last for more than six weeks.

Hill source of flow – having relatively low minimum flows compared with the high flows, and moderately frequent high flows of greater than about six times the low flow. Low flows typically occur during late summer/autumn and can last for 12–16 weeks, with high flows more common in late winter. Flood flows can have a very high magnitude (about 3,000 times the lowest flows).

Low elevation/spring source of flow – having relatively high minimum flows, a low to moderate frequency of low-intensity high-flow events (greater than about 1.5 times the low flows), with few high-magnitude flood events resulting in stable velocities and bed sediments.

For most river types, floods/freshes occur throughout the year and in most parts of New Zealand winter flows are generally higher than summer flows (this is reversed where rivers drain from the Southern Alps). The critical question for management becomes: is there a concomitant change in ecosystem structure and function that reflects these broad divisions in flow regimes? Are any biota/values *dependent* on these different flow regimes in a way that cannot be recognised or predicted, which would then justify promoting the retention of a 'natural flow regime'?

The relationship between the relative magnitude of natural minimum flows and the source of flow means that ecological flow requirements relative to natural flows will vary with the source of flow. Thus, the ecological flow requirements of rivers draining from hill sources are likely to be higher in terms of the natural minimum flow than those with mountain sources, with relatively more water available for allocation in rivers with mountain or spring sources than in rivers with hill sources.

New Zealand River Ecosystem Dependence on Flow Variability

While trout, native fish, invertebrates, and periphyton are all affected by flow variability to some extent, it appears that this is only at the extremes of intensity and frequency of events (low flows and floods). For benthic communities, many taxa such as the common mayfly *Deleatidium* spp. are able to survive and prosper under a variety of regimes – from spring-fed streams with almost no flow variability to the flashiest of mountain rivers (Quinn and Hickey 1990). Similarly, most common periphyton taxa (eg, *Ulothrix zonata*, *Gomphoneis herculeana*, *Spirogyra* spp.) live, and can prosper, across a similar range of flow regimes (Biggs and Price 1987; Biggs 1990). This indicates that they are well adapted to tolerate a range of flow conditions. Indeed, it appears that New Zealand aquatic systems (at least for invertebrates and periphyton) are characterised by populations that have evolved to be resilient and opportunistic, with flexible life-histories and (in broad terms) only poor specialisation to specific spatial or temporal habitats (Biggs 1990; Death and Winterbourn 1995; Thompson and Townsend 2000).

Artificially enhanced flow variability over daily scales, such as that which occurs in some hydro-electric controlled rivers, can be detrimental to species with lower mobility such as cased caddisflies due to drying effects (Irvine and Henriques 1984). Indeed, if high flows are too frequent, some biota will be unable to establish self-sustaining populations.

Aquatic macrophytes are only found in New Zealand waterways where floods are infrequent; thus they are dependent on low flow variability to prosper. Therefore, they tend to dominate benthic production in lakes or spring-fed rivers (Riis and Biggs 2003), and can become troublesome downstream of impoundments with constant flows (Biggs 1995).

Many common fish species also have flexible flow regime requirements. Brown trout, in particular, have very wide-ranging habitats from lakes and springs (ie, no flow variability) through to flashy mountain-fed rivers, although rainbow trout generally appear to be unable to establish viable populations in rivers with high flood disturbance unless there are downstream refuges such as lakes (Jowett 1990; Fausch et al. 2001). Native fish and brown trout seem to be better adapted to surviving large floods than rainbow trout (Jowett and Richardson 1989).

Spawning, egg hatching and migratory movements of some fish may be restricted to a few months of the year (McDowall 1995) and linked to the occurrence of suitable flow conditions. Early life stages are particularly vulnerable to high flows which can destroy entire year classes if they coincide with the egg or larval stage (Allen 1951; Hayes 1995). Species with asynchronous or extended periods of reproduction, such as upland bullies, will be influenced less by flow changes. They spread reproductive investment over an extended period which may be of adaptive value in unpredictable environments such as in New Zealand rivers (McDowall and Eldon 1997). Inanga (*Galaxias maculatus*) spawn on the banks of river estuaries on high spring tides and rely on subsequent inundation to stimulate hatching (McDowall 1990). If this inundation does not occur (eg, through high abstraction rates) then the spawning will fail.

Fish migration is often considered to be cued by flow variability. However, this appears uncommon in New Zealand rivers with their relatively frequent and short floods (an exception is salmon migration in shallow rivers). Studies of rainbow trout spawning migrations in the Tongariro River showed a weak, if any, link between flow and fish movement (Dedual and Jowett 1999; Venman and Dedual 2005). Floods and freshes in autumn carry larvae of some diadromous native fish to the sea, but this is largely opportunistic (Ots and Eldon 1975; Allibone and Caskey 2000; Charteris et al. 2003). Similarly, in some streams and rivers, floods in spring can open the mouth to the sea and allow juvenile diadromous fish to return to the river from the sea (Jowett et al. 2005). The timing of hydrological events can also have negative effects. Studies in the Kakanui River indicated that the adult trout population was regulated by variable recruitment and that in turn was associated with the occurrence of floods during spawning and incubation, with relatively small spring floods causing high mortality in emergent fry (Jowett 1995; Hayes 1995).

Flow variability in New Zealand rivers probably has its greatest impact on community structure and functioning. In streams with frequent floods, fish and invertebrates that are small and can colonise new areas rapidly, are often dominant (Scarsbrook and Townsend 1993). In such rivers, the periphyton community is usually sparse, with low species richness and diversity (Biggs 1990). In rivers with low flows and infrequent floods, communities are usually dominated by large, less mobile/more sessile, taxa such as filamentous green algae, macrophytes and snails (Biggs 1990; Quinn and Hickey 1990). Rivers with an intermediate frequency of bed-disturbing floods have been reported to have the highest diversity and biomass of benthic invertebrates (Townsend et al. 1995; Clausen and Biggs 1997), although other studies have not supported this conclusion (Death and Winterbourn 1995; Death 2002; Death and Zimmermann 2005).

Arguably, the most important requirement for flow variability is for the removal of accumulations of silt and periphyton on the river bed. Such accumulations can strongly degrade the quality of benthic habitats, but on flow-controlled (dammed) rivers can be dealt with by the use of well timed flow releases of the magnitude and frequency that is appropriate for the local reach channel geometry (Jowett and Biggs 2006). In some systems, stopping abstractions to develop high flows may be ineffective for significant cleansing: flows > 10 × baseflow will often be required.

Although flow variability is often thought of as an essential element of the flow regime that should be maintained, there is little published biological evidence that flow variability, in addition to uncontrolled floods, is essential for the maintenance of most instream values in New Zealand. Valued biological communities can be maintained in rivers where the flow regime has been extensively modified, but the needs of the instream values have been specifically identified and targeted in the management regime (which may include flushing flow releases) (Jowett and Biggs 2006).

The natural flow paradigm is a simple construct, based on the assumption that if you don't change the flow regime (and non-flow related factors also remain unchanged), the natural ecosystem will be maintained. Adoption of such an approach could place unnecessary restrictions on the use of water for out-of-stream purposes and may be suboptimal for the maintenance of key instream values. While some species may be adapted to a specific aspect of flow, this does not imply that the entire flow regime is necessary. This also doesn't allow for flexibility in habitat requirements and life-history strategies of biota that will enable them to cope with certain degrees of change. New Zealand flow regimes do differ according to climate and river type, yet the aquatic communities are broadly similar across these regimes. Flow regime decisions need to be guided by community values and the requisite ecosystem composition/functionality. Effort should be given to designing regimes that specifically support these values rather than relying on the nebulous objective of maintaining a 'natural flow regime' in the hope that the values will be protected. The selection of an appropriate flow regime for a river requires clear goals and targeted management objectives, with levels of protection set according to the relative values of the in- and out-of-stream resources. The challenge is to determine the aspects of the flow regime that are important for the various biota associated with their rivers, and to develop flow regimes that meet those needs – with appropriate monitoring to verify whether the biota responds as expected.

Habitat Requirements and Relationships with Abundance of Aquatic Fauna

It is the quality of the habitat that is provided by the flow that is important to stream biota, not the magnitude of the flow per se. In many streams, flows less than the naturally occurring low flow are able to provide good-quality habitat and sustain stream ecosystems. The flows that provide good habitat will vary with the requirements of the species and with the morphology of the stream; water velocity is probably the most important characteristic. Without it, the stream becomes a lake or pond. An average velocity of 0.3 m/s tends to provide for most stream life and will prevent the accumulation of fine sediment. Velocities lower than this are unsuitable for a number of fish species and stream insects and allow the development of nuisance growths of periphyton. In large rivers, water depth of more than 0.4 m provide habitat for adult brown trout, but in small streams depths in excess of 0.05 m are adequate for most stream insects and native fish. The flow at which these limiting conditions occur varies with stream morphology. Generally, minimum flow increases with stream size, because stream width increases with stream size. However, the relationship is not linear. Small streams require a higher proportion of the natural stream flow to maintain minimum habitat than do large streams.

Minimum flows do not necessarily influence fish populations nor are they the only factors controlling the fish population. Studies of trout in the Kakanui River showed that the total adult population was regulated by recruitment; that in turn was controlled by the occurrence of floods during spawning and incubation (Jowett 1995; Hayes 1995). Over the study period, low flows in the Kakanui River had no discernable effect on the trout population: lowest flow in the study period, 0.62 m^3/s, was a little higher than the MALF, 0.58 m^3/s.

Food availability may limit trout populations, as in the Horokiwi Stream (Allen 1951). Benthic invertebrate biomass was shown to be the most important factor relating to trout abundance in different rivers (Jowett 1992). In the Kakanui River the distribution of adult trout mirrored benthic invertebrate abundance, suggesting that it might be a limiting factor (Jowett 1995).

Less is known about the factors controlling native fish populations. Studies have been carried out to determine habitat preferences of native fish (eg, Jowett and Richardson 1995) and these have been independently verified by studies that show that native fish are more abundant where the average stream characteristics are close to the preferred habitat for the fish species (Jowett et al. 1996). Native fish densities are therefore often higher in small streams than in larger streams or rivers because the preferred habitat of native fish is usually for relatively shallow water. New Zealand native fish have evolved to cope with the conditions they experience in our rivers. Most galaxiids and eels are able to survive relatively long periods out of water and are capable of some overland movement. Many are also capable climbers and can penetrate to the headwaters of most rivers. Diadromous native species spend their early life stages in the ocean, thus avoiding the harsh riverine environment associated with frequent floods and freshes and unstable gravel substrate. Native fish live at densities of up to about $2/m^2$ in lowland areas, fish density reducing with elevation. The overwhelming influence of diadromy, the widespread distribution of the more common native species, and their well-defined preferences for relatively shallow water habitats, suggest that the total fish numbers and diversity will be controlled by diadromy, while instream habitat will control the distribution of fish within a river (Jowett and Richardson, 1996). Native fish distribution and abundance does not appear to be related to benthic invertebrate abundance. Flows that provide adequate native fish habitat are therefore likely to maintain native fish populations. Juvenile trout, like native fish, occupy shallow water and feed on smaller food items than adult trout. Their abundance was more closely related to the availability of food than to their habitat requirements, which are broad (Jowett et al. 1996). Predation can also limit native fish populations, as in Otago where many non-migratory galaxiid populations have been heavily impacted on by the introduction of trout (LePrieur et al. 2006).

Stream Size and Flow Requirements of Aquatic Communities

The composition of the fish community varies with stream size. Small streams are more suited to small fish than large, and vice versa. Small fish have lower swimming speeds and lower velocity and depth preferences than large fish. Adult salmonids usually move upstream or into tributaries to spawn and the juvenile fish rear in these areas, whereas the adults usually move back downstream to deeper waters after spawning. Because water depth and velocity generally increase with flow, there tends to be a flow that provides a maximum amount of habitat for a particular fish species and life stage. The amount of habitat (weighted usable area) at mean annual low flow in 71 New Zealand rivers was calculated for a range of fish species and life stages. When available habitat was plotted against flow and a smooth curve fitted, the peak of the curves gave an indication of the streams sizes that provided the most habitat for the species and life stages (Figure A2.1).

There is a general relationship between fish community, physical habitat requirements, and optimum size of river. Habitat increases with flow as streams become wider, until the stream reaches a size where further increases in stream size do not increase the amount of available habitat. The optimum size of a river for food producing habitat was about 15 m^3/s, for adult brown trout habitat 10 m^3/s, and the optimum size for trout fry/juvenile habitat (≤ 15 cm) was about 2 m^3/s. This is in agreement with general observations of the distribution of trout with adult trout in the larger streams and rivers, and trout rearing either in small streams or headwaters. The analysis can be extended to native fish and indicates that the optimum size of river for torrentfish, which are common in large braided rivers, is 10–15 m^3/s, whereas streams less than 1 m^3/s contain maximum physical habitat for many of the other native fish species.

A generalised analysis of habitat requirements (Jowett and Hayes 2004) produces similar results, with small streams suited to biota with low velocity and low depth requirements, and larger streams and rivers suited to larger species that prefer deeper water and higher velocities.

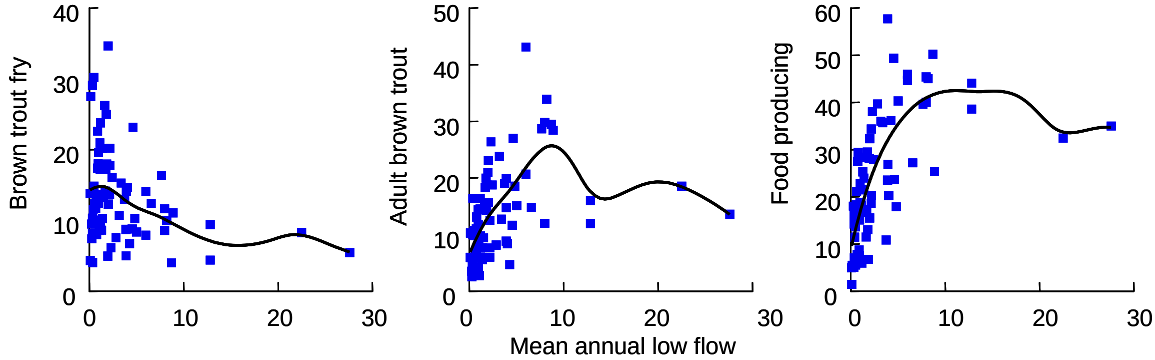

Figure A2.1: Weighted usable area (m^2/m) at mean annual minimum flow (m^3/s) in 71 New Zealand rivers, for brown trut and food-producing habitat.

Relative Importance of Flow Variability versus Minimum Flow

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

Wetland inundation may occur in a very specific flow band. For example the upper Taieri River breaches its river channel at a flow of around 10 m^3/s and begins to enter the scroll plain wetland (the area between the existing river and old meandering channels) and, at around 15–20 m^3/s the scroll plain is fully inundated. Therefore it is a relatively small flow band that is critical to maintaining this wetland and any reduction in the frequency of the occurrence of flows between 10 and 20 m^3/s would need to be investigated.

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

Large-scale projects like damming and major diversions will usually require detailed and specific studies to determine downstream flow requirements, such as minimum flows and their seasonal variation and flushing and channel-forming flows. Because minor diversions have little effect on floods and freshes, the main ecological concern is the minimum flow.

Flow variability and movement of bed sediments can have profound effects on stream ecosystems. Stable, spring-fed streams are subject to few floods, and the fish and plants that live in such streams are often unable to develop similarly or even to survive in less stable environments (Figure A2.2). On the other hand, gravel-bed rivers and their aquatic biota are in a constant state of change, caused by extreme flows (floods and droughts) and mobile bed sediments. Floods are the most important element of flow variability; flood frequency has been used in several biological models as the primary axis for classifying biological communities (Biggs et al. 1998). In streams with frequent floods, fish and invertebrates that are small and can colonise new areas rapidly are often dominant (Scarsbrook and Townsend 1993), and the periphyton community is usually sparse, with low species richness and diversity (Clausen and Biggs 1997; Biggs and Smith 2002). In streams with stable flow regimes, aquatic communities are thought to be influenced more by biological processes such as competition between species and grazing/predation than by external environmental factors (Poff and Ward 1989; Biggs et al. 1989).

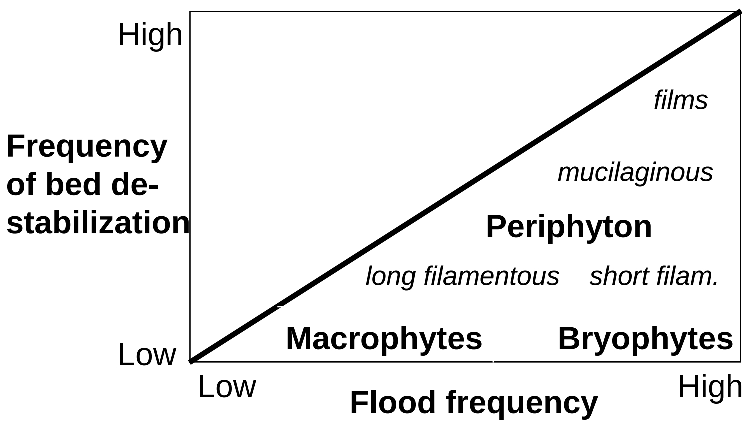

Figure A2.2: Effect of flow variability and substrate stability on river plants.

The biological effects of flow variability usually refer to the effects of floods or the effects of long periods of low flows (eg, Figure A2.3). However, we are not aware of any studies that demonstrate that small-scale flow variation is biologically important. In fact, frequent flow variations are usually considered detrimental. Daily and weekly flow fluctuations are often a feature of rivers downstream of hydropower stations. These fluctuations in flow create a 'varial' zone that is wetted and dried as water levels rise and fall. With frequent flow fluctuations, this zone will not sustain immobile plant and invertebrate species. Mobile species such as fish, and probably some invertebrate species, can make some use of this zone – especially for feeding in recently inundated areas of river bed, where there may have been some terrestrial invertebrates in the substrate. However, a varial zone that is wetted and dried at more frequent intervals than a week is unproductive and can be regarded as lost habitat.

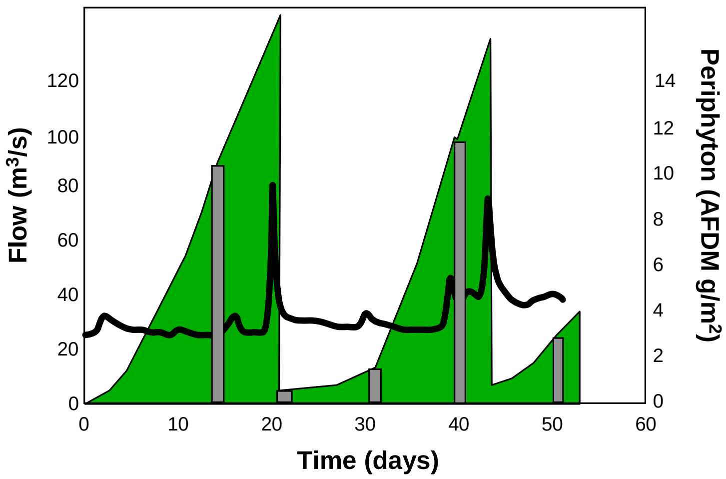

Figure A2.3: Effect of floods on periphyton accumulation in the Tongariro River (from Jowett and Biggs 1997).

It can be seen that determining the river flows required to maintain particular instream values may present significant challenges, particularly if there are several values that have different – or even opposite – requirements. Depending on specific proposals for use of the river – damming, large-scale run-of-river abstraction, minor abstractions, etc – it may be necessary to develop what might be called a 'designer flow regime', that considers the need to maintain floods, freshes, low flows, and aspects of flow variability. This, of course, means that the manager must have a clear idea of the outcomes that are desired, with regard to instream values, and the time and resources available to conduct an extensive ecological flow analysis.

▪ Appendix 3

Methods Used in Ecological Flow Assessments of Rivers

Methods Used in Ecological Flow Assessments of Rivers

Hydraulic Geometry Method

Hydraulic geometry methods *per se* are not widely used in New Zealand although hydraulic geometry is captured by hydraulic-habitat methods (eg, 1D hydraulic/habitat models). They predict how wetted area changes with flow, but do not have strong links to biological requirements. If hydraulic geometry is measured, the data can also be used for an analysis of habitat suitability. Although changes in wetted area provide more information than historical flow methods, the additional step of habitat analyses provides even more; this method should be considered part of the suite of hydraulic-habitat methods that use hydraulic geometry (eg, 1D hydraulic/habitat models, generalised habitat models, some regional models, WAIORA – see below).

Channel shape is determined primarily by geology and the flow regime of a river. The relationship between hydraulic geometry and flow can be defined between rivers or sites on rivers, using downstream hydraulic geometry or at a site methods; the latter is also known as at-a-station method. For alluvial rivers, downstream hydraulic geometry relationships between channel form and flow are similar in rivers worldwide (eg, Leopold and Maddock 1953; Kellerhals and Church 1989). River width increases with the square root of discharge (exponents range from 0.45 to 0.54: Park 1977; Kellerhals and Church 1989; Jowett 1998). Water depth and velocity also increase with discharge, although the relationships are not as well defined. At a site, hydraulic geometry relationships are more variable and less well reported. For New Zealand rivers, Jowett (1998) gives the average relationships at a site as:

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

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

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

where Q is the discharge, W the average width, D the average water depth, and V the average velocity.

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

When hydraulic geometry is used as a flow assessment method, the analysis is usually based on measurements of hydraulic data (wetted perimeter, width, depth or velocity) from one or several cross-sections in the stream. The aim of hydraulic methods is to maximise food production by keeping as much 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 from 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 2.3 in main text) 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 'inflection point' or more correctly, a breakpoint). If the wetted perimeter or width is used, the breakpoint is usually the point at which the water covers just the channel base. However, wetting of the channel base might not be enough to fulfil the requirements to depth and velocity for some species.

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

Habitat Methods

Of the three basic types of instream flow [incremental] methods (IFIM), historical flow methods are coarse and largely arbitrary, unless the natural flow paradigm is adopted and historical flows are specified so that they mimic natural flows. Hydraulic geometry methods provide information on the physical characteristics of the river, but do not have strong links to biological requirements. Habitat methods are an extension of the hydraulic methods. Their great strength is that they quantify the loss of habitat caused by changes in the natural flow regime, which helps the evaluation of alternative flow proposals. 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) describe IFIM as the "most appropriate for relative comparisons of habitat potential from among several alternative flow management proposals" and as "the method of choice when a stream is subject to significant regulation and the resource management objective is to protect the existing healthy instream resources by prescribing conditions necessary for no net loss of physical habitat". Nevertheless controversy has accompanied the development of the IFIM, in particular the hydraulic and habitat models (eg, physical habitat simulation (PHABSIM: eg, Mathur et al. 1985; Scott and Shirvell 1987; Kondolf et al. 2000; Hudson et al. 2003). A recent multi-authored review concluded with 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 (eg, floods). In other words, habitat methods can be used to set the 'outer envelope' of suitable living conditions for the target biota.

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

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

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

The two key elements of a habitat-based method are the habitat suitability criteria that are used to calculate habitat and the linkage between available habitat and aquatic populations. These two issues can be discussed and argued without resolution, although the bottom line is that there must always be suitable habitat if an aquatic species or use is to be maintained. An ecological justification can be argued for the MALF, and the concept of a low-flow habitat bottleneck for large brown trout has been partly justified by research (eg, Jowett 1992), but setting flows at lower levels, such as the 7-day, 5- or 10-year low flow ($Q_{7,5}$ or $Q_{7,10}$) is rather arbitrary. Hydraulic methods do not have a direct link with instream habitat; interpretation of ecological thresholds based on breakpoints or other characteristics of hydraulic parameters, such as wetted perimeter and mean velocity, are arbitrary and depend on rules of thumb and expert experience. On the other hand, habitat-based methods have a direct link to habitat use by aquatic species. They predict how habitat (as defined by various habitat suitability models) varies with flow and the shape of these characteristic curves provides the information that is used to assess flow requirements.

Habitat-based methods allow more flexibility than historical flow methods, offering the possibility of allocating more flow to out-of-stream uses while still maintaining instream habitat at levels acceptable to other stakeholders (ie, the method provides the necessary information for instream flow analysis and negotiation).

Generalised Habitat Models

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

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

Lamouroux and Capra (2002) proposed this model to reduce habitat survey effort but still retain much of the predictive power of conventional habitat-based models. These generalised habitat models use 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, although with some loss in precision (because they are based on average reach descriptions).

Tests of generalised models in France (Lamouroux and Capra 2002) and New Zealand (Lamouroux and Jowett 2005) found that habitat values for taxa were predictable from simplified hydraulic data. Reach hydraulic geometry (mean depth and mean width-discharge relationships), average bed particle size and mean 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 in France. The Reynolds number of reaches (discharge per unit width) governs changes in habitat value 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 habitat value in the different reaches. This is consistent with the preference of the benthic fauna, such as many of the native New Zealand fish species and benthic invertebrates, for riffles (Jowett and Richardson 1996; Jowett 2000), and the non-benthic aquatic fauna for runs or pools (eg, Jowett 2002).

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

Generalised models necessarily lose some information compared to conventional models such as river hydraulic habitat simulation (RHYHABSIM: Figure A3.1). This loss must be balanced against requirements for field work and experience in conventional modelling. In particular, hydraulic geometry relationships in reaches (required by generalised models) can be easily obtained from field measurements made at two different discharges or using regional models (Leopold et al. 1964; Jowett 1998; Lamouroux et al. 1998). By combining generalised models and hydraulic geometry relationships, estimating habitat values in multiple streams is possible from few field measurements; detailed topographies of stream reaches, associated velocity measurements and hydraulic model calibration are not required.

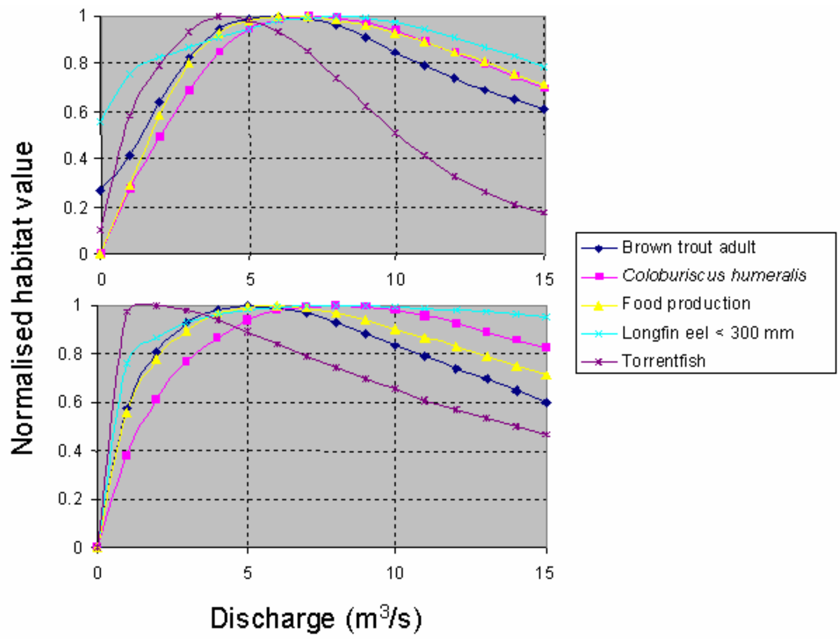

Figure A3.1: Comparison of normalised habitat per unit width predicted by habitat modelling in RHYHABSIM (upper) and the generalised method (lower).

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

Generalised habitat models are fitted to relationships between HV and width-standardised flow for a large dataset of rivers from throughout the nation or a region. Dividing flow by width standardises HV–flow relationships among rivers.

The generalised model takes the form:

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

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

The values of model coefficients for each taxa have been derived from a dataset of 99 reaches of New Zealand rivers. The reaches in this dataset have mean flows varying from 0.6 m^3/s to 53.8 m^3/s (the same data were used by Lamouroux and Jowett (2005). Jowett et al. (in press) describe generalised habitat models, and their derivation, more fully.

WAIORA – Generalised Habitat and Water Quality

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

Instream Flow Incremental Methodology

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

Habitat and Hydraulic Spatial Scales

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

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

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

Hydraulic Habitat Modelling Process

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

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

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

Hydraulic-habitat models can be separated into a hydraulic component and a habitat component. The hydraulic model predicts water velocity, depth and other hydraulic variables 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.

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

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

The hydraulic habitat analysis starts by choosing a particular species, size, and life stage and behaviour and defining suitability criteria. Waters (1976) proposed the use of a suitability index that varies between 0 (unsuitable) and 1 (optimal) as an alternative to binary criteria (0 unsuitable or 1 suitable) that had been used by in earlier hydraulic-habitat studies (McKinley 1957; Collings 1972). Intuitively, it seems reasonable to consider conditions that are of intermediate habitat value, between optimal and barely useful. For each cell in the grid (Figure A3.2), velocity, depth, substrate, and possibly other parameters (eg, cover) at the given flow are converted into suitability indices, one for each parameter. The suitability indices can then be combined (usually they are multiplied), and multiplied by the cell area to give an area of usable habitat. Finally, all the usable habitat cell areas can be summed to give the weighted usable area (WUA m^2/m) for the reach at the given flow. If the suitability is > 0 and ≤ 1, the cell will contribute to the total area, but if it is zero the cell makes no contribution. This whole procedure is then repeated for other flows to produce a graph of WUA versus flow for the given species. This graph has a typical shape, shown in Figure A3.3 with a rising part, a maximum and then may decline. The decline occurs when the velocity and/or depth exceed those preferred by the given species and life stage. Thus, in large rivers, the curve may predict that physical habitat will be at a maximum at flows less than naturally occurring.

The method is recognised as the most defensible for assessing instream flow needs in the United States although it has received some criticism (Mathur et al. 1985; Scott and Shirvell 1987). The fundamental criticism was that, although it seemed reasonable to assess instream flow needs on the basis of the amount of suitable habitat, there was no evidence that there was any correlation between species abundance and the amount of suitable habitat.

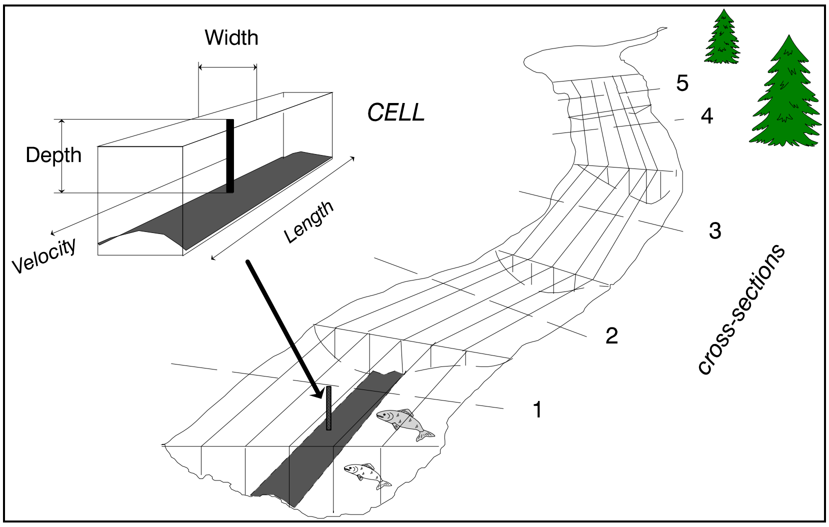

Figure A3.2: Habitat survey of a stream reach, showing the cell area represented by a point measurement.

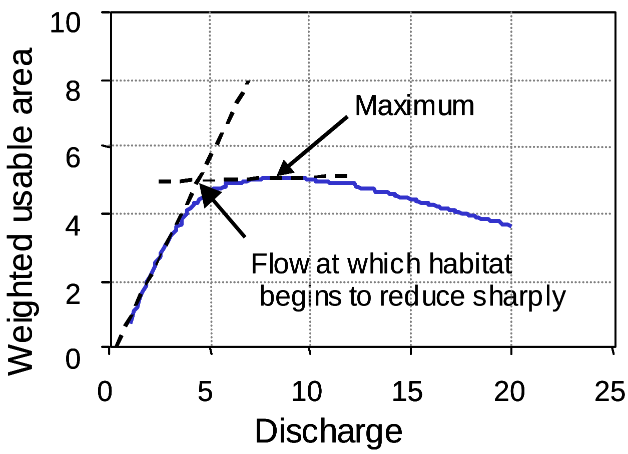

Figure A3.3: Selection of minimum flow at the point where habitat begins to decline sharply with decreasing flow.

Since then, some studies have demonstrated relationships between WUA and species abundance and in some instances – such as for benthic invertebrates – suitability is derived from species abundance and is correlated. However, the warning is valid and use of inappropriate habitat suitability curves could give misleading results. It is also necessary to consider all requirements for a species' continued survival. For example, the primary requirements for salmonids are both space and food (Chapman 1966), so assessment of instream flow needs for salmonids must consider both space and food requirements.

The relationship between habitat and flow (Figure 3.2) can be used to define a preferred flow range, a minimum flow, or a preferred maximum flow. As with hydraulic methods, the minimum flow can be defined as the break point or as the flow at which the habitat has dropped to a certain percentage of its value at mean or median flow. It can also be defined as the flow that has the lowest acceptable minimum amount of habitat in absolute terms. If minimum flows are at or above 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 A3.2) and the difference between maximum habitat and the amount of habitat at a high flow is relatively small. For example, most New Zealand native fish are found in shallow water along the edges of large rivers (Jowett and Richardson 1995) and there is usually some edge habitat available over a large range of flows.

When many fish species and life stages are present in a river, there are usually conflicting flow requirements. For example, young trout are found in water with low velocities, and adult trout are found in deep water with higher velocities. If the river has a large natural morphological variation with pools, runs and riffles, some of the different requirements may be provided for. Still, even in these rivers, and especially in rivers with small habitat variation, one species may benefit greatly from a reduction in depth and velocity, whereas habitat for another species will be reduced. If a river is to provide both rearing and adult trout habitat, there must be a compromise. One such compromise is to vary flows with the seasonal life stage requirements of spawning, rearing, and adult habitat; with the optimum flow gradually increasing as the fish grow and their food and velocity requirements increase. Biological flow requirements may be less in winter than summer because metabolic rates and food requirements reduce with water temperature. Whether fish are not food-limited in winter has not been tested in New Zealand and rarely has it been tested overseas. Some evidence has been found for reduced condition of trout in winter associated with reduced invertebrate food supplies (Filbert and Hawkins 1995; Simpkins and Hubert 2000). If flow requirements of individual species are different, solutions may be found by choosing one with intermediate requirements (Jowett and Richardson 1995) or by defining flow requirements for aquatic communities.

Generalised habitat models take into account the relationship between habitat and channel shape but do not require such detailed habitat surveys as a conventional instream habitat survey. Generalised models which are derived from 'WUA times flow responses from several rivers determined from traditional instream habitat surveys', can be applied to a specific stream knowing only the average width at one discharge.

Within the suite of habitat-based models, it is possible to select the model that is appropriate to the situation. In many situations, the simple generalised model, with one measurement of width and flow, can be used to define a minimum flow for the appropriate critical values and habitat retention levels. If the stream morphology is unusual (ie, substantially different from the range of rivers used to derive the generalised model) or if greater certainty is required, the width can be measured at two flows and WAIORA used to apply the generalised models. Finally, if the value of the instream or out-of-stream resource requires the most detailed level of consideration, instream habitat surveys and 1D, or even 2D, models can be used to predict habitat response curves for the critical values; or even fish energetics models, in the case of trout, which predict net rate of energy intake (Hayes et al. 2003; Kelly et al. 2005).