Simulating consequences of water allocation limits for the Auckland region

July 2012 Technical Report 2012/043

Auckland Council
Technical Report 2012/043
ISSN 2230-4525 (Print)
ISSN 2230-4533 (Online)

ISBN 978-1-927169-62-9 (Print)
ISBN 978-1-927169-63-6 (PDF)

Approved for Auckland Council publication by: Manager, Research, Investigations and Monitoring
Date: July 2012

Recommended citation

Franklin, P.A., Snelder, T.H., Diettrich, J.and Booker, D.J. (2012). Simulating consequences of allocation limits for the Auckland region. Prepared by the National Institute of Water and Atmospheric Research for Auckland Council. Auckland Council technical report, TR2012/043

Simulating consequences of water allocation limits for the Auckland region

Paul Franklin

Ton Snelder

Jani Diettrich

Doug Booker

National Institute of Water and Atmospheric Research Ltd

NIWA Client Report No: HAM2012-119
NIWA Project: ARC12505

Executive summary

Auckland Council (AC) is currently developing a Unitary Plan that will establish default limits for the allocation of surface water across the Auckland region. The objective of these limits is to provide a specific level of environmental protection, while enabling out-of-channel water use at specified levels of availability and reliability. To achieve this, limits must include at least minimum flows (the flow below which no water can be abstracted) and total allocation (the total quantity of water that can be abstracted upstream of any location).

Default limits can be defined using "rules of thumb" that are based on hydrological indices, such as the mean annual low flow (MALF). This is the approach that is taken by the proposed National Environmental Standard for Ecological Flows and Water Levels (proposed NES; MfE 2008). Rules of thumb are easily applied, but there are two main disadvantages. Firstly, the consequences for both environmental values and out-of-channel water users (i.e., availability and reliability) are not clearly articulated, making justification of the rules difficult. Secondly, the consequences are spatially variable because flow regimes and the relationship between environmental protection and flow are spatially variable.

The Environmental Flows Strategic Allocation Platform (EFSAP) provides a method to evaluate the consequences of setting different water resource use limits across all parts of a catchment or region, including those for which detailed information is not available. It integrates scientific tools to enable the concurrent evaluation of consequences for instream habitat and reliability of supply for out-of-channel water uses, accounting for the interaction between the flow regime, minimum flow and total allocation limits at all locations.

In this study we used EFSAP to simulate the consequences of various potential sets of limits (i.e., minimum flows and total allocations) for all river and stream reaches in the Auckland region with a mean flow greater than 10 L s^{-1}. A range of alternative scenarios, both more environmentally conservative and more resource use enabling than the proposed NES rules, were simulated.

The indicator species selected for analysis were banded kokopu (Galaxias fasciatus), shortfin eel (Anguilla australis), longfin eel (Anguilla dieffenbachia) and common bully (Gobiomorphus cotidianus). These were selected based on their presence and value in the Auckland region. EFSAP is based on the analysis of flow duration curves (FDCs). In this study we used the annual FDC and the March FDC to represent the average consequences and the consequences under the most restrictive summer conditions respectively.

Under the proposed NES default limits for small rivers (minimum flow 90 per cent MALF, total allocation 30 per cent MALF) the spatial patterns of reliability at the management flow (where partial restrictions begin) are similar for both the annual and March FDCs. For the annual FDC, overall variability throughout the region is relatively low, with the majority of locations having reliability of between 80 per cent and 95 per cent at the management flow. However, for the March FDC, while the spatial patterns are similar, the magnitude of variation between locations in the region is much greater.

Overall regional variability in reliability at the minimum flow (i.e., where abstractions must cease) was low for the annual FDC with most locations having a reliability of between 90 per cent and 95 per cent. For the March FDC, regional variability was greater, with a range for the majority of locations of 70 per cent to 100 per cent reliability, but the median reliability was similar between both at 93.4 per cent for the March FDC and 94.5 per cent for the annual FDC.

The spatial pattern of consequences for instream physical habitat was similar for all four indicator species that were modelled. The median loss of habitat for the annual FDC was -6.0-6.5 per cent for banded kokopu, shortfin eels and longfin eels, and slightly higher at -8.2 per cent for common bullies. For the March FDC, the median loss of habitat was -7.0 per cent for common bullies and -5.0-5.5 per cent for the other three indicator species.

To improve equitability for stakeholders, we attempted to define spatially discrete management units with relatively uniform outcomes for each of the values. However, no significant differences were found between classes based on geology, land cover, stream order or river size. Consequently, it was concluded that there was insufficient spatial differentiation at the regional scale being considered to justify the definition of different spatial management units within the region.

A range of allocation scenarios were simulated for the Auckland region for both the annual and March FDCs. The consequences for each value for the full range of simulated scenarios were then summarised in a decision space diagram that encompasses minimum flows ranging from 10 per cent to 100 per cent of MALF and total allocation limits that range from 10 per cent to 150 per cent of MALF, each in 10 per cent increments.

Once objectives have been set for each value, the decision space diagrams can be used to determine which combination of limits satisfies each objective. Once the subset of limits that satisfy the objective for each individual value have been defined, they can be combined to find the set of limits which meet all objectives. In some cases, the defined objectives for all values will result in a combination of limit options that overlap. Water resource managers therefore have the choice of defining limits that satisfy all objectives. However, in some circumstances there will be no combination of limits that satisfies all objectives. In this situation, a compromise has to be found between the different values until an acceptable combination of limits can be agreed upon. The decision space diagrams can assist in this trade-off process by illustrating to stakeholders and resource managers how limits interact with each other and the relative consequences of alternative management decisions.

It must also be recognised that EFSAP does not evaluate all values that may be relevant for a given location. It also does not explicitly consider flow variability, or the temporal sequencing on flows. It is also based on the assumption that instream physical habitat at low flows is limiting. These factors must therefore be considered when determining the most appropriate combination of limits. However, despite these limitations, EFSAP provides a robust and defensible approach to evaluating the relative merits of different combinations of limits and therefore will allow AC to more transparently communicate and set water resource limits that meet their nominated objectives.

Table of contents

1.0 Introduction 8

1.1 Background 8

1.2 Scope 8

2.0 Limits for water allocation 10

3.0 Methods 13

3.1 EFSAP model description 13

3.2 Applying EFSAP in the Auckland region 17

4.0 Results 21

4.1 Proposed NES rules 21

4.2 Spatial patterns 30

4.3 Decision space diagrams 30

5.0 Discussion 37

5.1 Consequences of proposed NES default limits 37

5.2 Use of decision space figures for limit setting 38

5.3 Limitations 40

6.0 Conclusions 43

7.0 Acknowledgements 44

8.0 References 45

Appendix A 48

Reviewed by: Approved for release by:

Ned Norton Helen Rouse

1.0 Introduction

1.1 Background

The National Policy Statement for Freshwater Management (MfE 2011) requires that regional councils define environmental flow limits that include both minimum flows and total allocation limits. Auckland Council (AC) is developing a Unitary Plan that, among other things, will establish default limits for the management of abstraction of surface water. The Unitary Plan will define interim (or default) limits for surface allocation water across the Auckland region. Interim limits may be altered in future subject to more detailed studies. The objective of these limits is to provide a specific level of environmental protection, while enabling use of water at specified levels of availability and reliability. To achieve this, limits need to include at least minimum flows (the flow below which no water can be abstracted) and total allocation (the total quantity of water that can be abstracted upstream of any location).

One method for setting default limits is to use “rules of thumb” that are based on hydrological indices such as the mean annual low flow (MALF). This is the approach that is taken by the proposed National Environmental Standard for Ecological Flows and Water Levels (proposed NES). The proposed interim limits are based on a proportion of MALF, which varies for two river size classes. Small and large rivers are those with mean flows less than and greater than 5 m^3s^{-1} respectively, and minimum flows are set at 90 per cent and 80 per cent of MALF respectively. The proposed NES sets total allocation for small and large rivers at 30 per cent and 50 per cent of MALF respectively (MfE 2008).

Rules of thumb based on MALF are easily applied to set minimum flows and total allocation, but there are two disadvantages. First, the consequences of retaining any specific proportion of MALF are not clearly articulated, making justification of the rules difficult. Second, the consequences of uniform limits to define minimum flows and total allocation are spatially variable, because flow regimes and the relationship between environmental protection and flow are variable in space. Simulation modelling of potential water allocation limits will enable AC to evaluate and choose among options for minimum flows and total allocations by assessing the consequences for environmental protection, and the availability and reliability of water for out-of-stream use across the Auckland region.

1.2 Scope

The scope of this project involved the use of the Environmental Flows Strategic Allocation Platform (EFSAP) to simulate the consequences of various scenarios for water allocation for all river and stream reaches of interest (i.e., all river and stream locations where surface water takes occur or could occur) in the Auckland region. EFSAP is a software program developed by NIWA to enable assessment of different environmental flows and total allocations on reliability of supply and instream habitat. There are two key steps in undertaking this work: a) testing and improvement of hydrological estimates; and b) simulating the consequences of various environmental flow and allocation scenarios. The details involved in these steps are discussed below.

1.2.1 **Hydrological estimates**

The EFSAP program requires estimates of hydrological characteristics for all locations of interest. The first step in the project was to obtain the best available estimates for these hydrological characteristics for the Auckland region, as well as estimates of their uncertainties. A separate report (Booker & Woods 2012) details the findings of that part of the study and includes a comparison of methods for calculating hydrological indices, from purely empirical methods (i.e., statistical modelling) to those applying more physically-based approaches.

1.2.2 **Simulation analyses**

The aim of this project was to undertake a series of simulation analyses of different water allocation scenarios using EFSAP. The output from these analyses describes the retention of physical habitat for target species, the availability of water for out-of-channel uses (i.e., the estimated total allocation as a flow in $m^3 s^{-1}$), and the reliability of that water supply (proportion of the time abstractions are fully or partially restricted). Because these outputs are spatially variable, the results were to be presented as maps and also summarized statistically (e.g., histograms showing the distribution of values or catchment averages of the output values).

Firstly, the consequences of the proposed NES minimum flow and total allocation rules on reliability of supply and instream habitat for target species were to be simulated for all rivers and streams in the region. The target species on which to base the analysis of physical habitat were to be defined in consultation with AC staff based on observed data and values identified in the Unitary Plan and other documents. The consequences for more than one target species were to be simulated. A range of alternative scenarios, ranging from more environmentally conservative limits than the proposed NES to limits more permissive of resource use than the proposed NES, were also to be simulated.

It was anticipated that rules may need to vary spatially in order to account for differences in hydrological regimes and physical habitat-flow relationships. Ideally, rules would be identified for a small number of easily defined sub-regions or river classes (e.g., based on stream order or dominant catchment geology), such that the consequences were reasonably uniform within and between such management units.

2.0 Limits for water allocation

Limits to water resource use are typically applied for two reasons. First, limits are imposed to constrain human-induced alteration of river flows to levels that are judged to be sufficient to sustain environmental values (Acreman et al. 2008, Poff et al. 2010, Richter et al. 2006). Second, economic considerations require that limits on water resource use are specified so that the total availability of water resources and their reliability is quantified and understood.

In situations where water resource development is intense (e.g., involving storage and diversions), many aspects of a river's flow regime may be altered, including high flows and the seasonal distribution of flows. In these situations, complex limits commensurate with the environmental impact and scale of the development are required to manage the effects. However, the more common situation is multiple small run-of-river or groundwater abstractions that are spread throughout a catchment with little, if any, facility for water storage. Environmental effects in this situation arise primarily due to the cumulative aggregation of abstractions, which can result in increased frequency and duration of low flows, but usually have a negligible impact on higher flows (Nilsson & Renöfält 2008). In these situations, the impact of abstraction can be controlled by defining two types of limit; minimum flows (the flow at which abstractions must be restricted to protect river ecosystem characteristics and functions) and total allocation (the total volume or rate that water can be abstracted).

The minimum flow and total allocation interact with each other and the flow regime to determine the consequences for environmental protection (i.e., the magnitude and duration of low flows) and out-of-channel uses (i.e., the proportion of the time abstractions are partially and fully restricted). Because of these interactions, limits are only effective, and the consequences of those limits for instream values and out-of-channel use can only be established, if both minimum flow and total allocation are defined.

An example of how the minimum flow and total allocation interact with the flow regime to determine consequences for instream and out-of-channel use at a site is illustrated with reference to the proposed NES rules. The top panel of the diagram shown in Figure 1 shows how a measure of environmental state, the availability of instream habitat for adult brown trout (Salmo trutta), varies with flow at a site. The mean flow at the site is $11.7 m^3 s^{-1}$ and the MALF is $2.6 m^3 s^{-1}$ (Point A, Figure 1). The availability of suitable physical habitat for brown trout declines with both increasing and decreasing flow from an optimum at $6.2 m^3 s^{-1}$ (Point B, Figure 1). The lower panel shows a flow duration curve (FDC) for the natural flow regime (i.e., the FDC that would have occurred if no abstractions were occurring).

Under the proposed NES, the minimum flow at the site would be 80 per cent of MALF or $2.1 m^3 s^{-1}$, (Point C, Figure 1), which equates to retention of 87 per cent of the physical habitat available for adult brown trout at the MALF (Point D, Figure 1) (i.e., a decline of 13 per cent relative to the amount of habitat available under average natural low flow conditions). The minimum flow and the natural FDC (i.e., the FDC that would have occurred with no abstractions) can be used to define the proportion of time that complete restriction (cessation of abstractions) occurs. Complete restriction occurs for the proportion of time that the natural flow is at or below the minimum flow, equating to 2.4 per cent of the time in this case (Point E, Figure 1).

The management flow is defined by adding the total allocation to the minimum flow, and describes the flow below which partial restriction of abstractions would be required to maintain the river at the minimum flow. Under the proposed NES, total allocation at the example site would be 50 per cent of MALF or $1.3 m^3 s^{-1}$, resulting in a management flow of $3.4 m^3 s^{-1}$ (Point F, Figure 1). Thus, when natural flows are below $3.4 m^3 s^{-1}$ (i.e., 10.0 per cent of the time, point G, Figure 1), the allowable abstraction is restricted to the natural flow minus the minimum flow. A consequence of this restriction is that residual flows (natural flows minus abstractions) are constant at the minimum flow (i.e., “flat-lining”) for the proportion of time that natural flows are between the management and minimum flows (i.e., between points G and E, Figure 1). Thus, in this example the river would be flat-lined 7.6 per cent of the time.

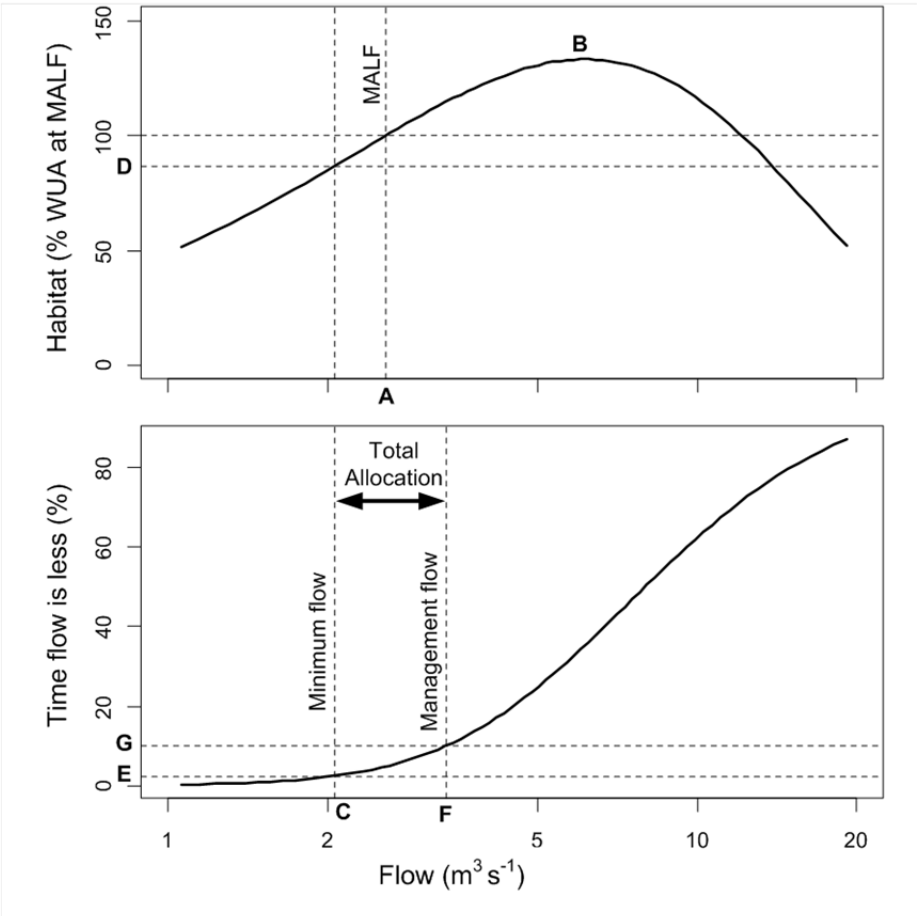


Figure 1: Schematic diagram of aspects of limit setting based on flow duration curve and flow-habitat relationships. The two plots are for a river with a mean flow of $11.7 m^3 s^{-1}$ and a MALF of $2.6 m^3 s^{-1}$. The top plot shows the flow-habitat relationship for adult brown trout estimated using generalized instream habitat models (Jowett et al. 2008) using the method described by Snelder et al. (2011). The lower plot shows the flow duration curve that was estimated using methods described by (Snelder et al. 2011). The lower plot shows the nominated minimum flow, management flow and proportion of the time that abstractions would be restricted (partially and completely). Letters indicate key values that are referred to in the text.

This example demonstrates that the choice of limits involves a trade-off between values. Choosing a lower minimum flow would reduce protection of environmental values, but increase reliability for out-of-channel use, whilst choosing a higher allocation limit would increase the availability of water for out-of-channel uses, but with lower reliability and reduced protection for environmental values. It also illustrates that when arbitrary rules, such as the proposed NES, are used to define limits, this trade-off is pre-determined.

Ideally, limits should not be prescribed by arbitrary rules, but instead be based on environmental and water resource use objectives. If the objectives are clear (i.e., defined levels of protection for environmental values and availability and reliability of water for out-of-channel users), decision-making is more transparent. The role of scientific tools is to provide defensible criteria that will meet these objectives. A significant scientific challenge to this process is the integration of tools in a way that accounts for how minimum flows, total allocation and the flow regime interact, to define a relevant range of options for objectives and associated limits. When setting water resource use limits over broad geographical areas, a further challenge is spatial variation in environmental characteristics. Spatial variation means that relevant objectives are likely to vary within and between catchments, and that the consequences of any set of limits are likely to be variable. This means that objectives and associated limits often need to be spatially specific (i.e. applying to the particular set of circumstances in a geographically defined location; Snelder & Hughey 2005, Snelder et al. 2004).

3.0 Methods

3.1 EFSAP model description

EFSAP is a tool to enable planners and water allocation decision-makers to simulate and compare spatially explicit water management scenarios at catchment, regional and national scales. It is able to simulate the spatially explicit consequences of multiple takes on both out-of-stream and in-stream values, demonstrate the trade-off between environmental state and resource use, and allow comparison of different water allocation management scenarios. It is based on the application of generalized models across all locations in a spatial framework. Further details of the model structure are described below.

3.1.1 Spatial framework

The spatial framework for EFSAP is the River Environment Classification (REC; Snelder & Biggs 2002), which comprises a digital representation of the New Zealand river network and a classification system that are contained within a Geographic Information System (GIS). The river network representing the Auckland region comprises approximately 6360 segments with a mean length of ~785m. Each segment is associated with several attributes including the total catchment area, stream order, and the climatic, topographic, geological, and land-cover characteristics of the upstream catchment. The REC classifies all river and stream segments into classes at several levels of detail (Snelder & Biggs 2002). The first and second levels of the REC assign individual segments of the river network into classes that discriminate variation in the climate and topography of the catchment. Within the Auckland region there is negligible variation at the first and second levels of the REC. The third and fourth levels of the REC are based on variations in catchment geology and land-cover, which do display spatial variation within the Auckland region.

3.1.2 Hydrological data

EFSAP requires estimates of several hydrological characteristics including: MALF, mean flow (Q_{bar}), and the shape of the FDC. FDCs are a hydrological tool that is used to represent the percentage of time flows are equalled or exceeded for a particular river location (Vogel & Fennessey 1995) (Figure 2). This project required both annual (i.e., calculated across the entire year) and monthly (i.e., calculated for individual months) FDCs so that the consequences for availability and reliability of water supply for out-of-channel uses could be reported for the whole year and the most restrictive (summer) months. Approaches for estimating these hydrological characteristics are described by Booker and Woods (2012). The methods with the lowest uncertainties have been used with EFSAP to undertake the simulation analyses for this project.

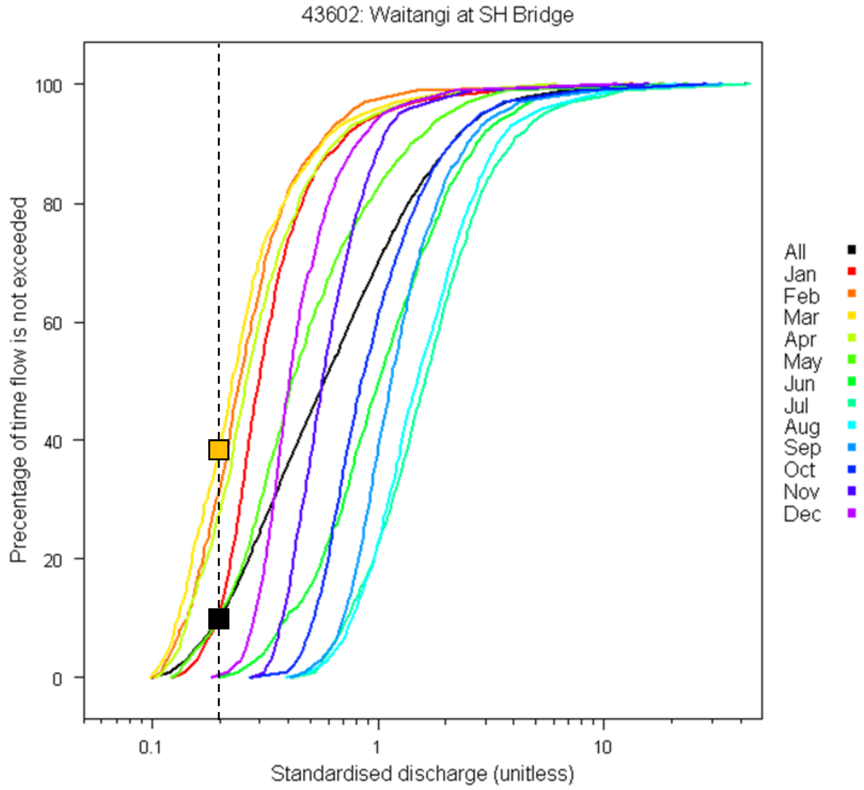


Figure 2: An example of annual and monthly FDCs for a network segment in Auckland. Minimum flow is indicated by the vertical dashed line. Reliability for any given month is indicated by the point at which the FDC meets the dashed line. For example, the black square indicates reliability based on the annual FDC (c. 90%), the orange square indicates reliability based on the March FDC (c. 60%).

3.1.3 **Generalized habitat v. flow relationships**

EFSAP utilizes coupled generalized models of mean wetted width versus flow and habitat versus reach-averaged specific discharge (width/flow) to describe the relationship between habitat availability and flow at a site.

3.1.3.1 Estimating hydraulic geometry

National estimates of at-station hydraulic geometry parameters are provided by Booker (2010). Booker (2010) defines a power-law relationship between discharge, Q (m^3s^{-1}), and mean wetted width, W (m), for each river reach:

$$\log(W) = d_0 + d_1 \log(Q) + d_2 (\log(Q))^2 \quad (1)$$

$$d_0 = a_0 + a_1 \log(A) \quad (a)$$

$$d_1 = b_0 + b_1 \log(A) \quad (b)$$

$$d_2 = c_0 + c_1 \log(A) \quad (c)$$

where A is catchment area (km^2) and a, b, and c take values dependent on REC classes (Table 1). These models are used in EFSAP to estimate the mean wetted width and subsequently to compute width-flow relationships for all REC network segments.

3.1.3.2 Estimating instream physical habitat

Conventional instream physical habitat models link hydraulic model predictions with microhabitat-suitability criteria to predict the availability of suitable habitat at various discharge rates (e.g. RHYHABSIM; Clausen et al. 2004, Jowett 1996, Jowett & Biggs 2006). The availability of suitable physical habitat is commonly expressed as Weighted Usable Area (WUA) in m^2 per 1000 m of river channel (Figure 3). WUA is an aggregate measure of physical habitat quality and quantity, and will be specific to a particular discharge and taxa/life stage. Instream physical habitat models can be used to assess WUA over a range of flows and therefore to make predictions of how habitat changes with changes in flow.

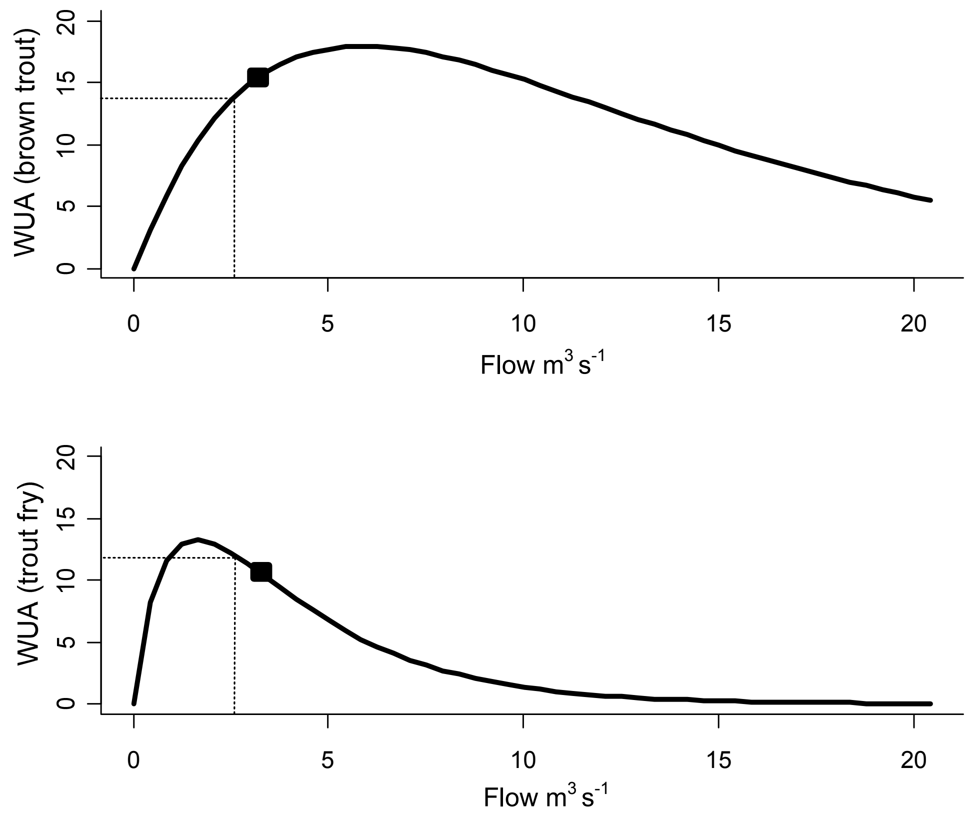


Figure 3: WUA versus flow curves for a network segment (mean flow = 20 $m^3 s^{-1}$) for adult brown trout and brown trout fry. These curves were defined by combining equations 1 and 2. MALF for the segment (3.3 $m^3 s^{-1}$) is shown by the black square on the curve. WUA at the proposed NES minimum flow of 80% MALF are shown by the dashed lines. Note that WUA decreases between MALF and the minimum flow for adult brown trout but increases for brown trout fry.

Criticisms of instream physical habitat models include a lack of biological realism (Orth 1987) and failure of microhabitat-suitability criteria to reflect the detailed mechanisms that lead to density–environment associations (Booker et al. 2004, Lancaster & Downes 2010, Mathur et al. 1985). However, many microhabitat suitability models have a high degree of transferability between rivers and are therefore useful bases for the physical management of stream catchments (Lamouroux et al. 2010). The models have been applied throughout New Zealand (Lamouroux & Jowett 2005) and the world (Dunbar & Acreman 2001), primarily to assess impacts of abstraction. PHABSIM in particular has become a legal requirement for many impact studies in the USA (Reiser et al. 1989) and a standard tool employed to define minimum flows in New Zealand.

Generalized instream habitat models (Lamouroux & Jowett 2005) have been developed from the results of many individual habitat studies conducted throughout New Zealand. These models generalize the relationship between flow and habitat in natural stream reaches based on simple reach-average hydraulic characteristics (Lamouroux & Jowett 2005). Therefore, when linked with hydraulic geometry models, generalized habitat models make it possible to simulate the relationship between flow and habitat over whole river networks (see examples in Jowett 1998, Lamouroux 2008, Lamouroux & Capra 2002, Snelder et al. 2011). We used the generalized instream habitat models provided by (Jowett et al. 2008) to estimate WUA as a function of reach-averaged specific discharge (width/flow). The flow-habitat relationships describe a unimodal shape that depends on two coefficients, *j* and *k* that are specific to a taxa and *i*, which is specific to a reach:

$$WUA = i\left(\frac{Q^{j}}{W^{j-1}}\right)e^{-k\left(\frac{Q}{W}\right)} \qquad (2)$$

The ratio of WUA at two discharge rates depends only on discharge rates and the width-discharge relationship, but not on the reach coefficient *i*. Consequently, the width-flow relationship (Equation 1) can be combined with Equation 2 to estimate the change in habitat with changes in flow over a whole river network (Lamouroux & Souchon 2002).

3.1.4 **Analysis options**

EFSAP is based on the analysis and simulation of four key variables:

- Flow changes (c.f. total allocation) (ΔQ)
- Minimum flow (Q_min)
- Reliability (R)
- Habitat change (ΔH)

When undertaking a simulation, any two of these variables may be specified and the other two will be calculated at all locations on the river network. For example, to simulate the consequences of the proposed NES minimum flow and total allocation limits for small rivers (<5 $m^3 s^{-1}$), flow change (ΔQ) would be set as 30 per cent MALF and minimum flow (Q_min) as 90 per cent MALF, and reliability of supply (R) and habitat change (ΔH) for the target species would be calculated by the model for all locations.

EFSAP can be run in two modes: global and local. Global simulations are used to evaluate the spatial consequences of uniform rules or objectives across the river network. In this mode, all reaches are treated as independent and thus the spatial distribution of takes is not taken into consideration and effects are not accumulated down the river network. The global mode was used for this project. The local mode allows simulation of the cumulative effects of site specific takes. In this mode, the location, take volume (ΔQ) and minimum flow (Q_min) of every abstraction is specified and the effects are accumulated down the river network. This approach is more suitable for more detailed, catchment specific investigations where good data are available on the location of takes.

3.2 Applying EFSAP in the Auckland region

3.2.1 **Assumptions**

This project takes a regional approach to simulating the consequences of different water allocation limits. The models upon which EFSAP is based are not calibrated for every position on the river network, but instead provide a generalised estimate that, when considered collectively, help to understand regional scale patterns. Results should therefore be evaluated and interpreted at a regional scale, and should not be used for assessments at specific locations on the river network.

Booker and Woods (2012) showed that characteristics of the FDC can vary between months, and monthly FDCs are different to the annual FDC. This means that for a given minimum flow and allocation limit, reliability of supply for out-of-channel uses will vary between months, with the lowest reliability occurring in the month with the greatest frequency of low flows. To allow for this variability, EFSAP simulations for Auckland have been run using the overall annual FDC and the FDC for March only. The March FDC was chosen for analysis because, on average, the greatest frequency of low flows occurs in this month and therefore it is the most resource limiting month.

Due to the uncertainties associated with predicting FDCs in urban areas, it was agreed with AC that all REC reaches with an urban landcover class would be removed from the analysis. All stream reaches with a mean flow of less than 10 L s^{-1} were also removed from the analysis due to the practicalities of applying limits in such small streams and to increase model efficiency.

Estimates of reliability of supply were based on the position of various proportions of the 7-day MALF on the flow duration curves (Booker & Woods 2012). For this calculation, we had a choice of three different estimates of MALF to locate on the flow duration curve at each location. These were: a) MALF from HUC (Hydrology of Ungauged Catchments projects); b) specific MALF estimated from a random forest regression model and then multiplied by catchment area; and c) the flow on each estimated flow duration curve that corresponded to the estimated position of MALF (as predicted by a random forest model of the proportion of time for which MALF is exceeded) on that flow duration curve (Booker & Woods 2012). Given that there may be errors associated with both estimated MALF and estimated FDCs, we chose to apply the last of these three options, as it was the most likely to produce accurate estimates of reliability of supply. Comparisons of reliability of supply at 100 per cent of MALF (results not shown) showed that there was little difference between our second and third methods of estimating MALF, but that the HUC method produced higher estimates of reliability of supply.

When simulating consequences for environmental state and reliability for out-of-channel uses, it is assumed that the full quantity of allocated water available is taken all of the time. This represents the worst case scenario. It is recognised that in reality this is rarely the case, but greatest demand for out-of-channel uses typically occurs when the resource is most limited (i.e., dry summers) and therefore it is important that water resource use limits are designed to provide sufficient protection of environmental values and reliability of supply at full capacity.

EFSAP uses instream physical habitat as its measure of environmental state. The use of physical habitat is based on the assumption that habitat availability, rather than other factors such as water quality or migration barriers, is the primary limiting factor on the target species. Physical habitat is used as a surrogate for the suitability of a site to support the target species, but the availability of suitable habitat does not mean that a species will be present, and the quantity of suitable habitat does not necessarily correlate with species abundance.

3.2.2 **Indicator species**

Generalised habitat models are currently only available for a restricted number of species and life stages in New Zealand (Appendix A, Table 3). The values for the model coefficients were derived by Jowett et al. (2008) from a dataset of 99 stream reaches in New Zealand. The 'flow demand' (in terms of optimal discharge per unit width; Appendix A, Table 3) for some species is logical based on our understanding of the traits of the individual species, e.g., torrentfish (which prefer fast flowing riffle habitats) having the highest demand of the native fish species. However, the optimal discharge defined by the Jowett et al. (2008) models are less intuitively logical for others, e.g., common bully (which have very plastic habitat requirements, but relatively high flow demand). It is possible that this is symptomatic of a sampling bias in the data used to derive the models towards daytime habitats in wadeable gravel rivers. Further work is required to validate the use of these models, and particularly their transferability across different river types. This research would help to reduce uncertainty in the models and their output. It would also be beneficial to expand the range of species and life stages included to provide more flexibility in selecting relevant target species.

The indicator species used for this assessment were determined with reference to both known (New Zealand Freshwater Fish Database; NZFFD) and predicted fish distributions (Leathwick et al. 2008), and values identified in local planning documents for the Auckland region (Table 1).

Table 1: Indicator species used for EFSAP simulations in Auckland.

Indicator species	Justification
Longfin eel (<30 cm)	Cultural value and conservation status
Shortfin eel (<30 cm)	Cultural value and broad distribution
Banded kokopu (juvenile)	Characteristic species of more natural streams
Common bully	Broad distribution

3.2.3 Scenarios

Uniform rules can be defined as standardised, unvarying limits that apply equally to all sites in a class. The proposed NES default minimum flow and allocation limits are uniform rules, divided into two classes based on river size (small < mean flow $5m^3s^{-1}$ ≥ large). Uniform rules are easy to apply and manage, but can result spatially varying consequences for both instream and out-of-channel water uses and consequent equity problems for stakeholders.

We first simulated the consequences of the proposed NES default minimum flow and allocation limits for small (Q_Min 90% MALF, ΔQ 30% MALF) rivers. We then simulated a range of scenarios encompassing both more environmentally conservative and more resource enabling limits than the proposed NES defaults. All scenarios were based on proportions of MALF (Q_min 10-100% MALF; ΔQ 10-150% MALF in 5% increments) and were applied to all sites in the catchments of interest.

3.2.4 Analyses

Reliability of supply was determined for both the proportion of time that abstractions are partially restricted (Point G, Figure 1) and the proportion of time that no abstraction is possible because natural flows are at or below the minimum flow (Point E, Figure 1). These two points were termed ‘reliability at the management flow’ and ‘reliability at the minimum flow’ respectively.

The availability of physical habitat was described in terms of the weighted usable area (WUA). For every segment of the river network, we calculated WUA at MALF and at the scenario minimum flow for each species. To allow comparison of all network segments we expressed the WUA at the minimum flow as a percentage of the WUA at MALF (Point D, Figure 1).

We analysed the spatial patterns of the consequences for reliability and habitat under the proposed NES scenarios. Results were initially mapped and a visual inspection used to identify likely drivers of any spatial differentiation in consequences. We then used REC reach attribute data, e.g., stream order or catchment geology, and other attribute data provided by Auckland Council, e.g., geology and natural stream management areas, to try and distinguish spatial differences and subsequently define spatial classes or potential “management units” with relatively uniform consequences for reliability and habitat. If significant spatial differences could be identified and defined, results were split into spatial classes for further analysis, with the consequences for the full range of uniform rule scenarios summarised statistically and presented as a ‘decision space’. This allows visual comparison of the range of potential outcomes for habitat and reliability resulting from different combinations of minimum flow and allocation limit.

4.0 Results

4.1 Proposed NES rules

The proposed NES allocation rules for small rivers (minimum flow 90 per cent of MALF and total allocation 30 per cent of MALF) were applied to all locations in the Auckland region and used to evaluate the spatial patterns and variability in consequences for resource reliability and instream habitat for the target species. Scenarios were run using both the annual and March FDCs to compare the consequences under average annual and low flow conditions.

The spatial patterns of reliability at the management flow are similar for both the annual and March FDCs, i.e., those locations with low reliability under the annual FDC also have low reliability under the March FDC and vice versa (Figure 4). For the annual FDC, overall variability throughout the region is relatively low, with the majority of locations having reliability at the management flow of between 80 per cent and 95 per cent under the proposed NES small river rules (Figure 10). However, for the March FDC, while the spatial patterns are similar, the magnitude of variation between locations in the region is much greater, with reliability at the management flow ranging from about 50 per cent to 100 per cent and the median is lower at 84 per cent (Figure 10).

The reliability of supply at minimum flow, which represents the proportion of time that at least some water is available for allocation, has a different spatial pattern to reliability at the management flow. For the annual FDC, there is a general west to east gradient of declining reliability (Figure 5), which most likely reflects prevailing climatic influences. However, this regional pattern is not discernible in the results based on the March FDC (Figure 5), with patterns instead more closely resembling those observed in the reliability at management flow (Figure 4). Overall regional variability in reliability at the minimum flow was low for the annual FDC (most locations between 90 per cent and 95 per cent) (Figure 10). For the March FDC, regional variability was greater, with a range for the majority of locations of 70 per cent to 100 per cent, but the median reliability was similar between both at 93.4 per cent for the March FDC and 94.5 per cent for the annual FDC.

The spatial pattern of consequences for instream physical habitat was similar for all four indicator species that were modelled (Figures 6 to 9). Overall regional variation was generally between 0 per cent and 20 per cent loss of habitat relative to MALF for all four species, for both the annual and March FDCs. For all four species, the consequences for habitat were smaller and displayed lower regional variability for the March FDC, relative to the annual FDC (Figure 11). The median loss of habitat for the annual FDC was -6.0-6.5 per cent for banded kokopu, shortfin eels and longfin eels, and slightly higher at -8.2 per cent for common bullies. For the March FDC, the median loss of habitat was -7.0 per cent for common bullies and -5.0-5.5 per cent for the other three indicator species.

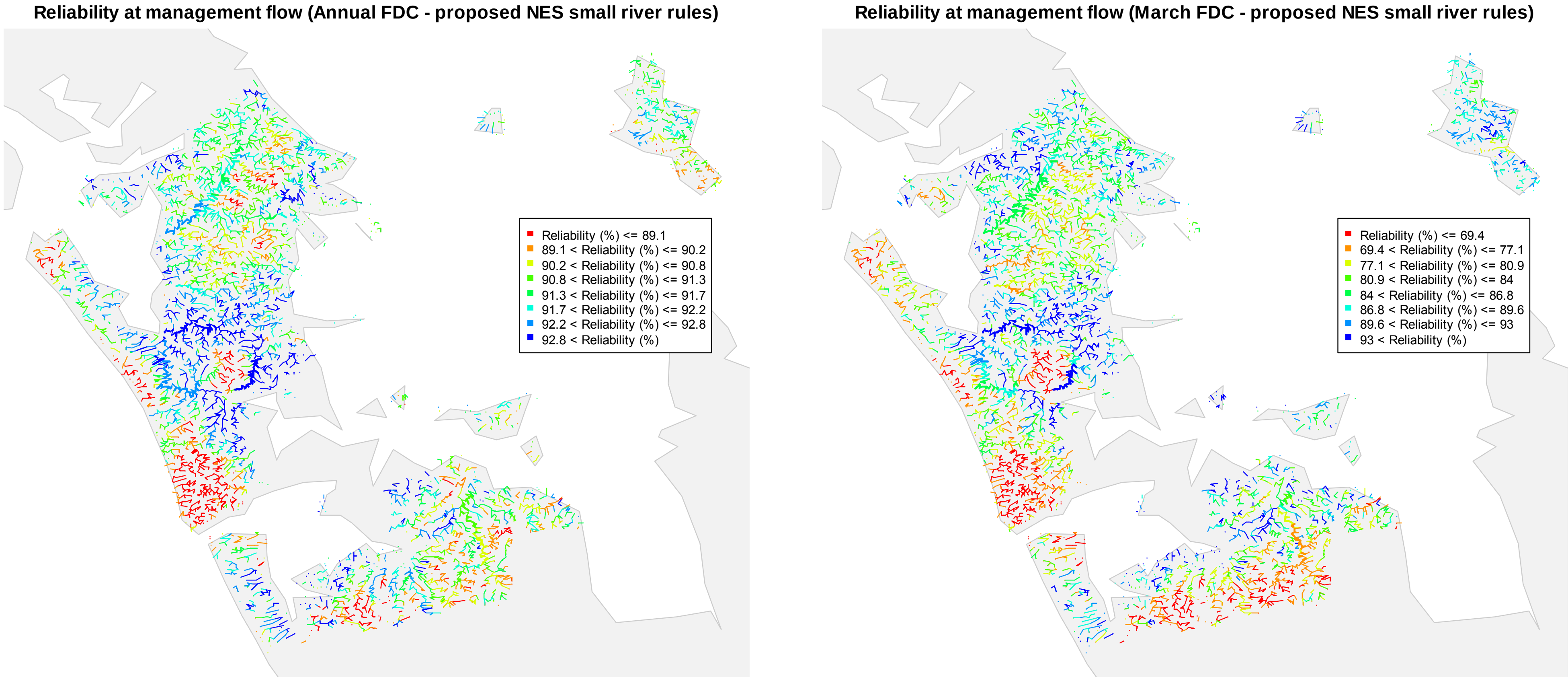


Figure 4: Maps showing the reliability at management flow in the Auckland region under the proposed NES small river rules. Streams in urban catchments have been removed from the analysis.

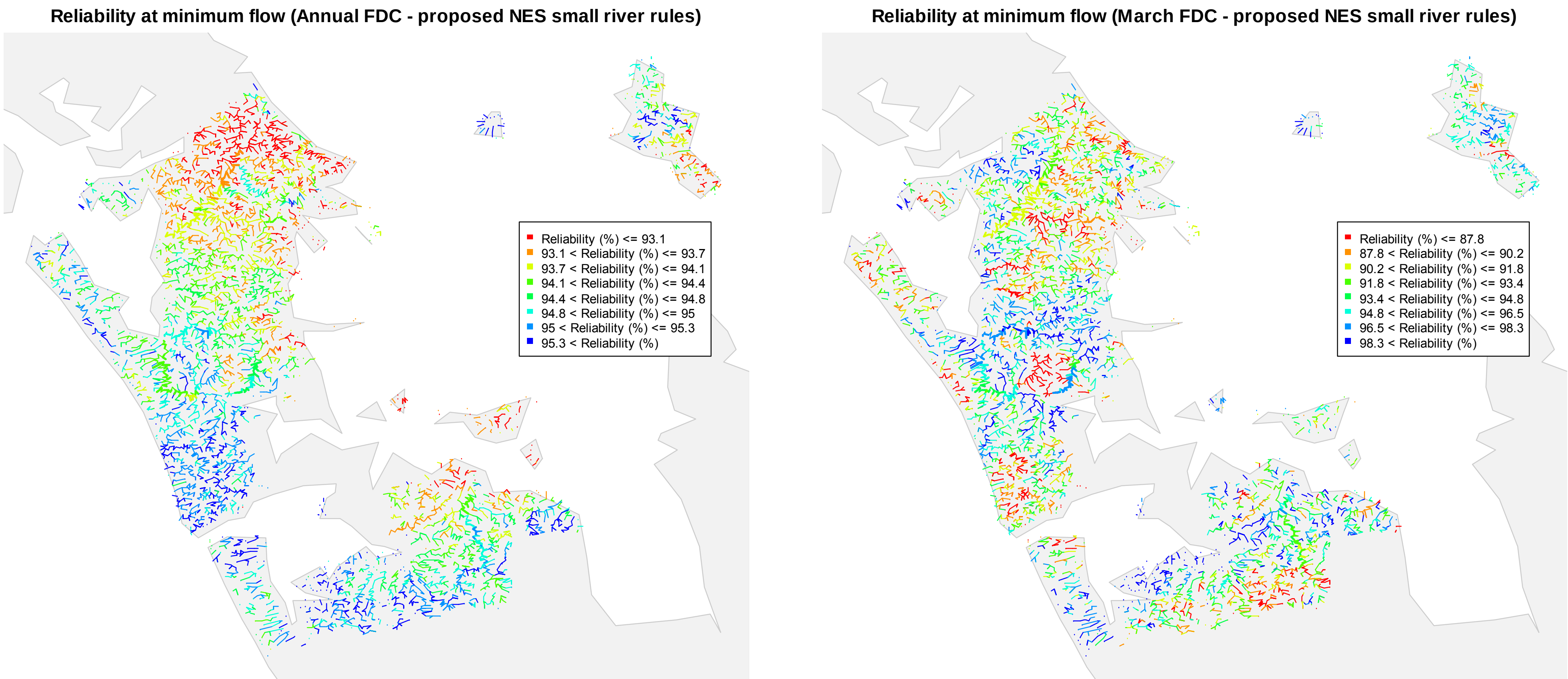
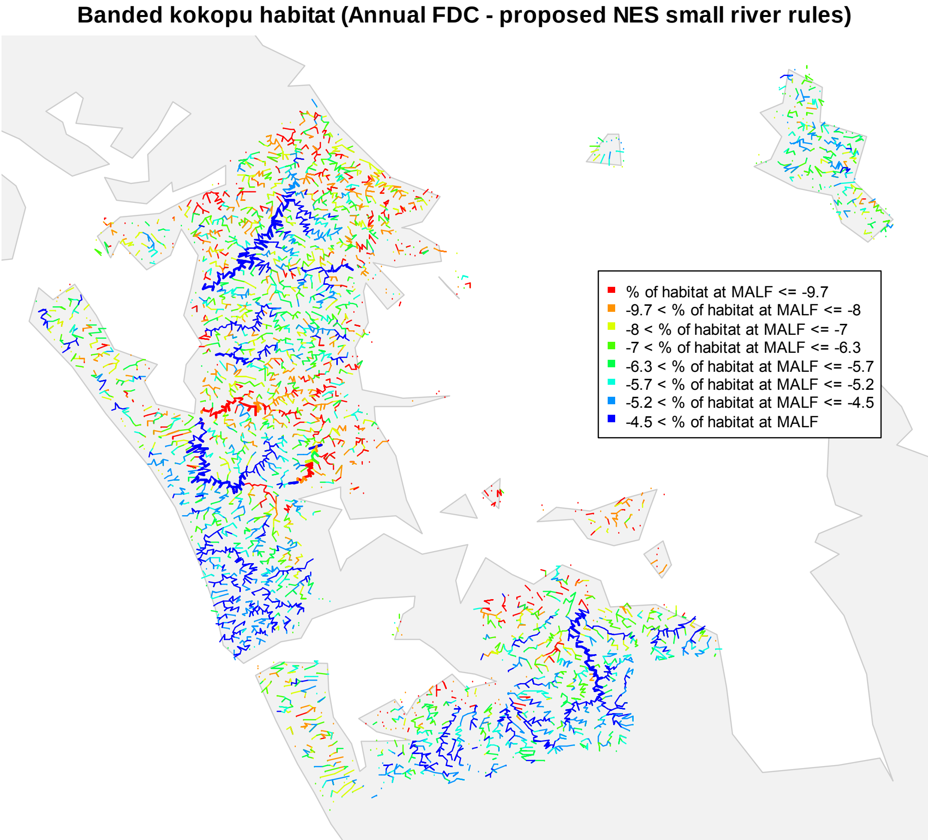


Figure 5: Maps showing the reliability at minimum flow in the Auckland region under the proposed NES small river rules. Streams in urban catchments have been removed from the analysis.



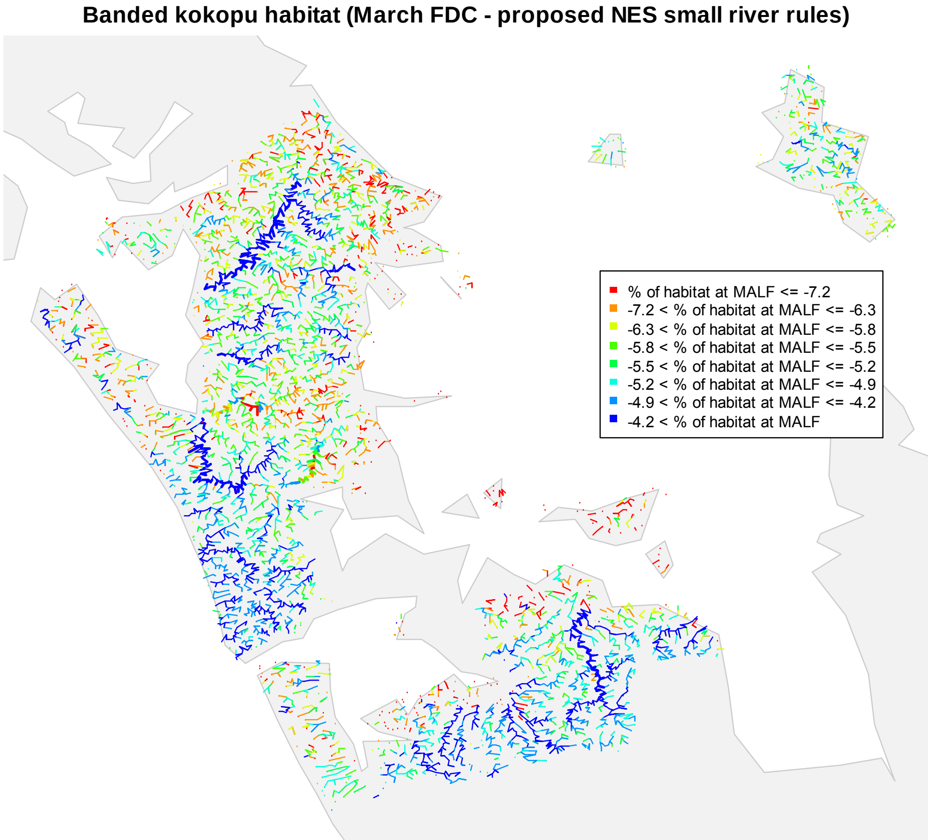


Figure 6: Maps showing the change in habitat for banded kokopu in the Auckland region under the proposed NES small river rules. Streams in urban catchments have been removed from the analysis.

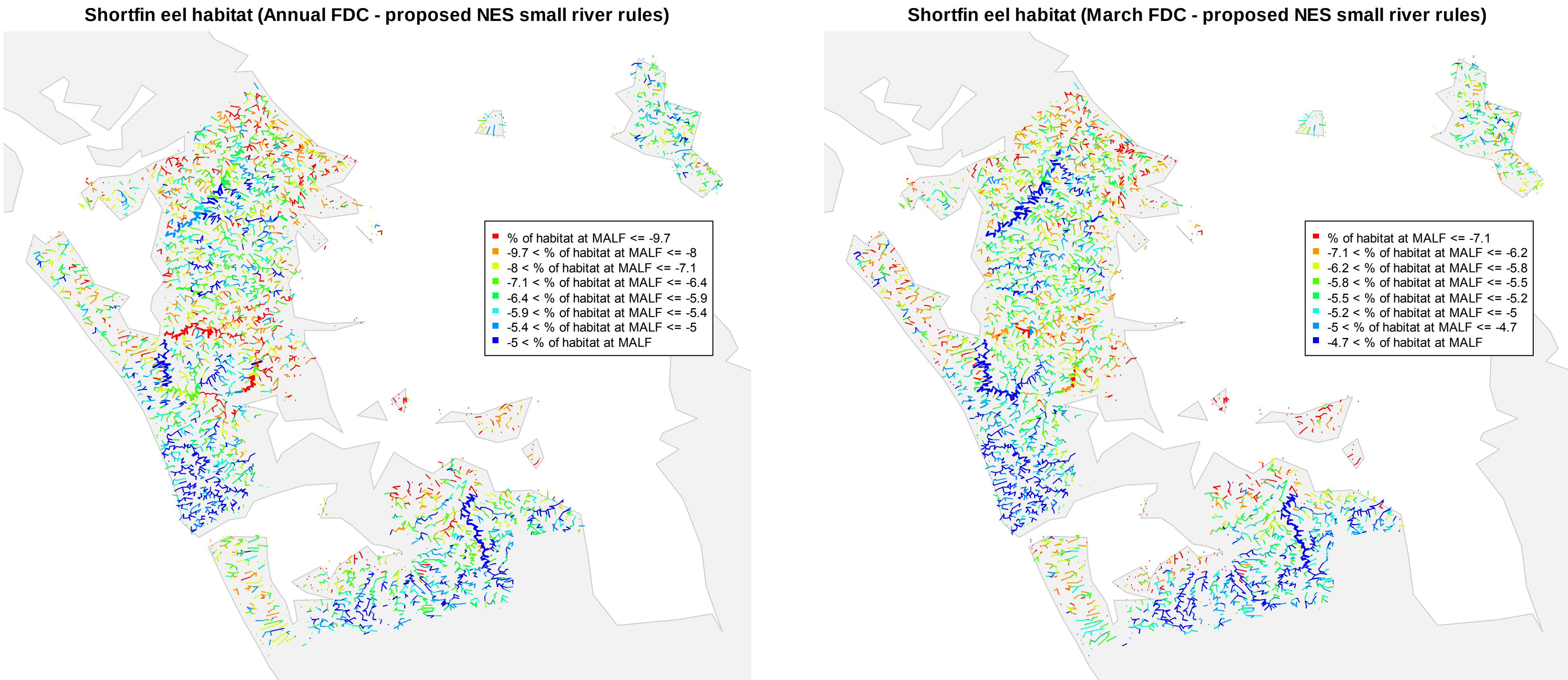


Figure 7: Maps showing the change in habitat for shortfin eel in the Auckland region under the proposed NES small river rules. Streams in urban catchments have been removed from the analysis.

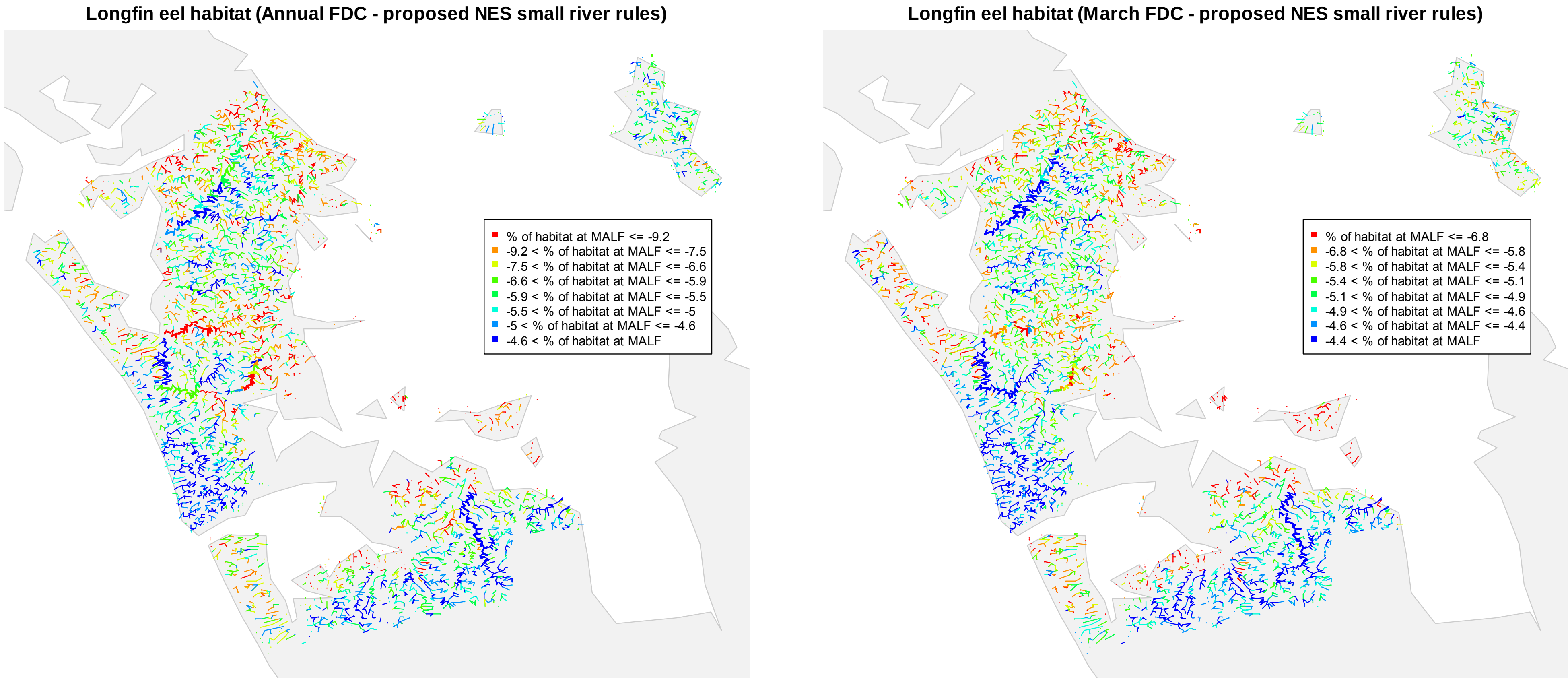
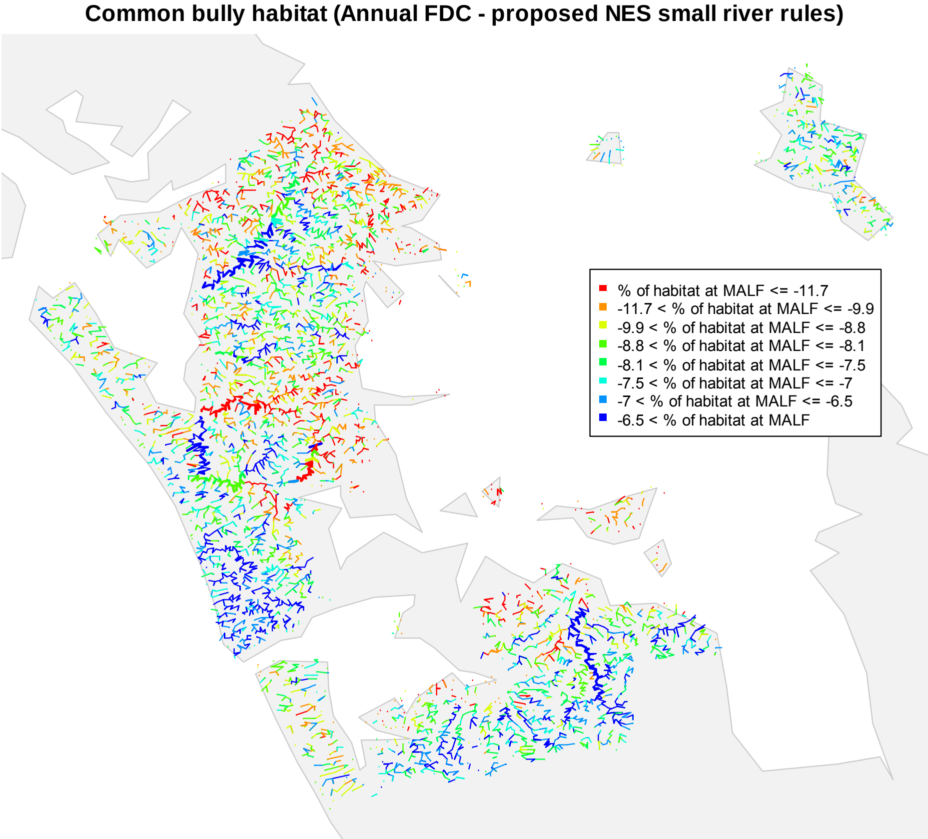


Figure 8: Maps showing the change in habitat for longfin eel in the Auckland region under the proposed NES small river rules. Streams in urban catchments have been removed from the analysis.



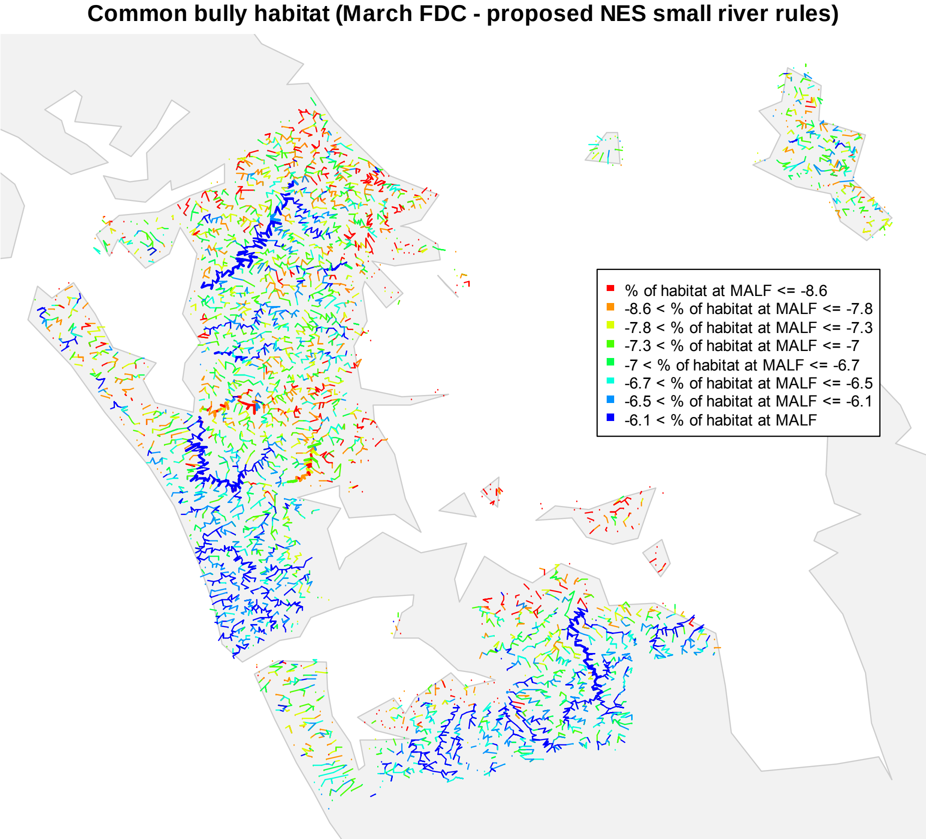


Figure 9: Maps showing the change in habitat for common bully in the Auckland region under the proposed NES small river rules. Streams in urban catchments have been removed from the analysis.

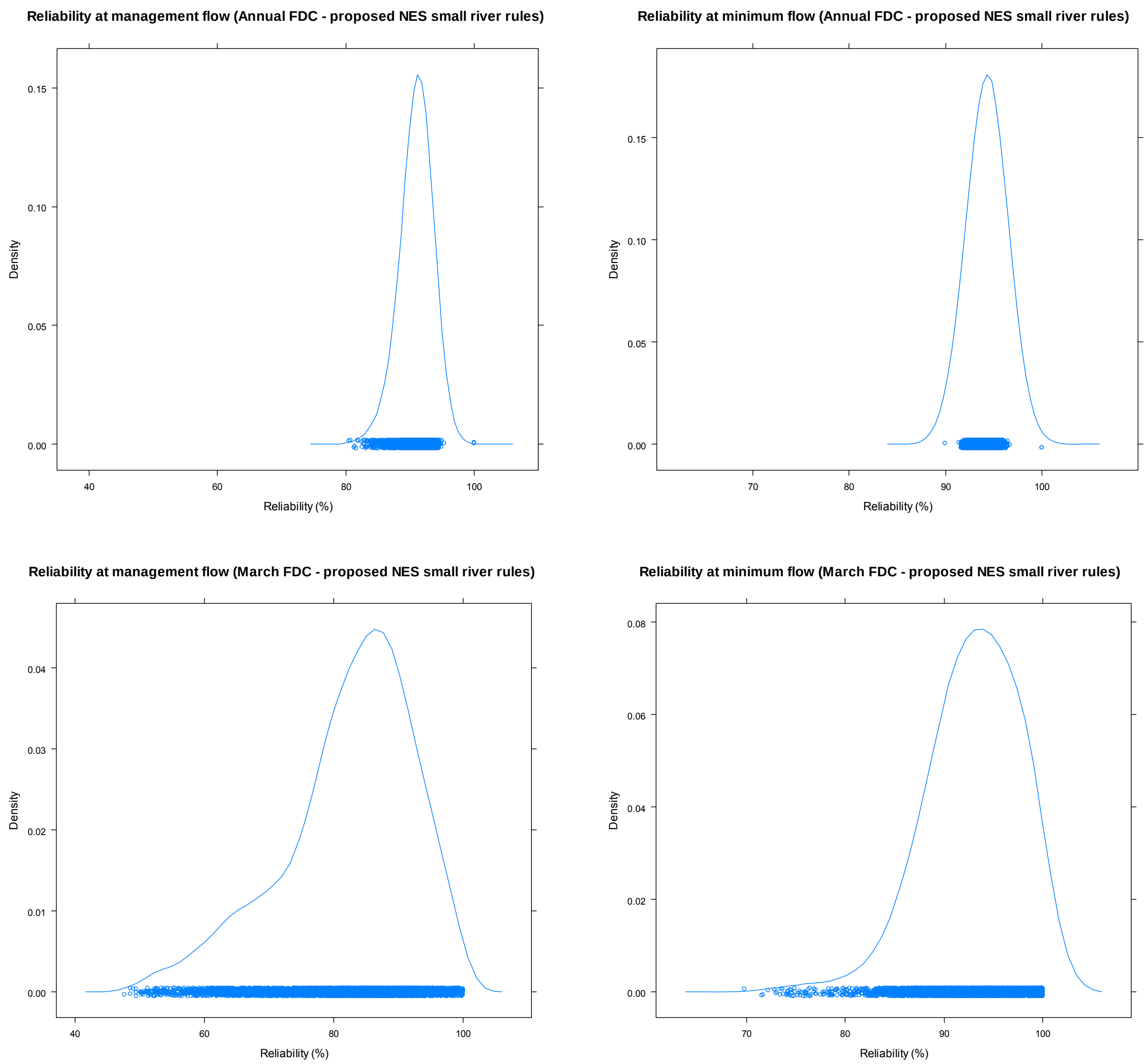


Figure 10: Variation in reliability at management and minimum flows across all locations. Top: Annual flow duration curve; Bottom: March flow duration curve.

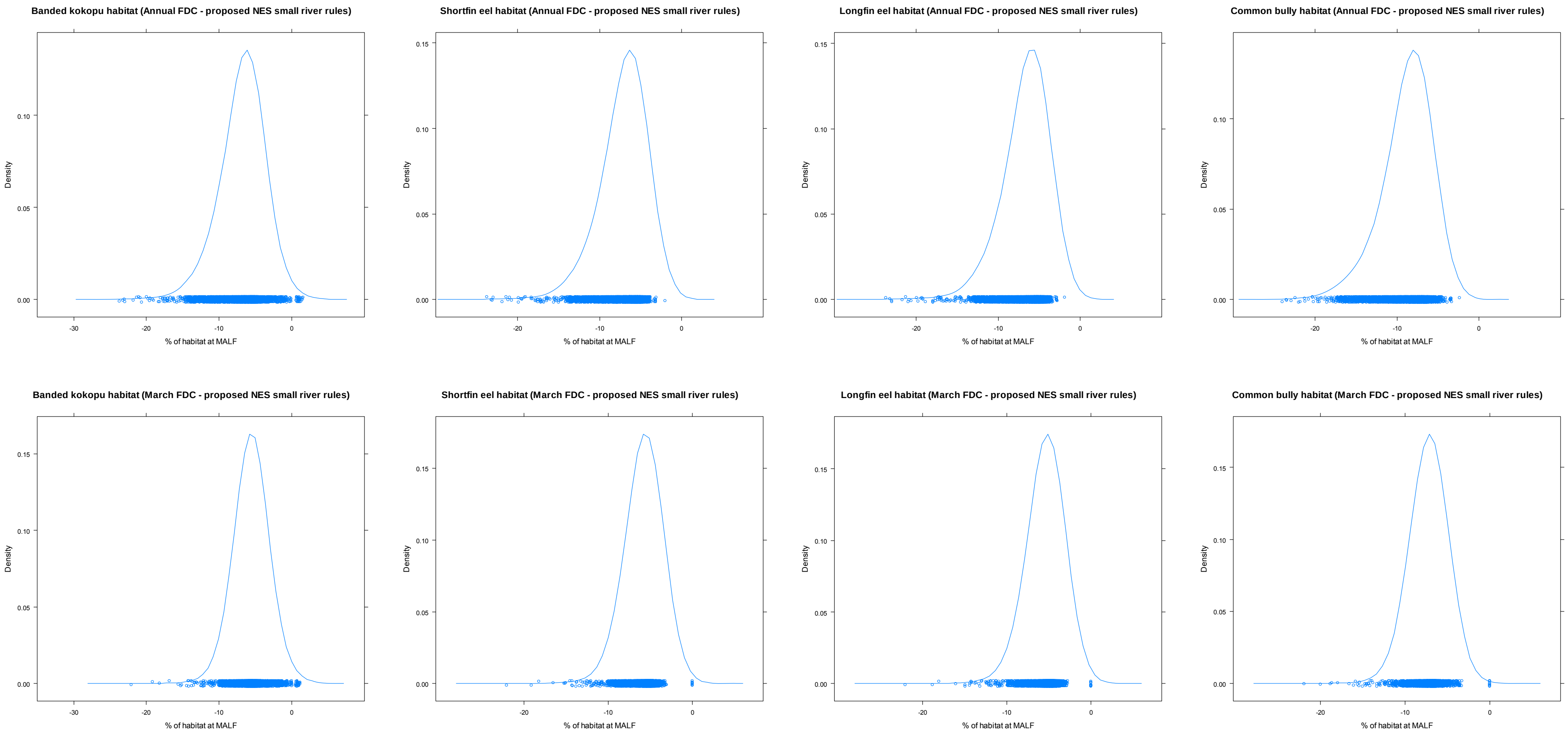


Figure 11: Variation in habitat response across all locations for the four indicator species. Top: Annual flow duration curve; Bottom: March flow duration curve.

4.2 Spatial patterns

To improve equitability for stakeholders, we attempted to define spatially discrete management units with relatively uniform outcomes for each of the values. Exploratory data analysis was carried out using REC classes and AC geology and natural stream management areas (NSMAs) to differentiate spatial groupings.

Analyses showed that for the annual FDC no significant differences could be identified in consequences for reliability between REC stream order, geology or landcover classes, AC geology classes, or for NSMAs. There were also no significant spatial differences in consequences for instream habitat for the four indicator species using the same spatial classes.

For the March FDC, slight differences in reliability were identified between some of the REC landcover and AC geology classes. However, further analysis of these results showed that these differences separated out less than 3 per cent of locations across the whole region. These locations were also widely spread geographically. No differences were distinguished for instream habitat. Consequently, it was concluded that there was insufficient spatial differentiation at the regional scale being considered to justify the definition of different spatial management units.

4.3 Decision space diagrams

A range of allocation scenarios were simulated for the Auckland region using EFSAP, for both the annual and March FDCs. The consequences for each value for the full range of simulated scenarios were then summarised in a decision space diagram. Here we have presented decision space diagrams that encompass minimum flows ranging from 10 per cent to 100 per cent of MALF, and total allocation limits that range from 10 per cent to 150 per cent of MALF, each in 10 per cent increments. The full range of scenarios presented may not be considered acceptable in all circumstances and this should be considered as part of the limit selection process.

The decision space diagrams for reliability at management flow, reliability at minimum flow and for instream physical habitat for the four indicator species can be found in Figures 12, 13 and 14 to 17 respectively. Guidance on how these diagrams can be used to inform the limit setting process can be found in the discussion (Section 5.2).

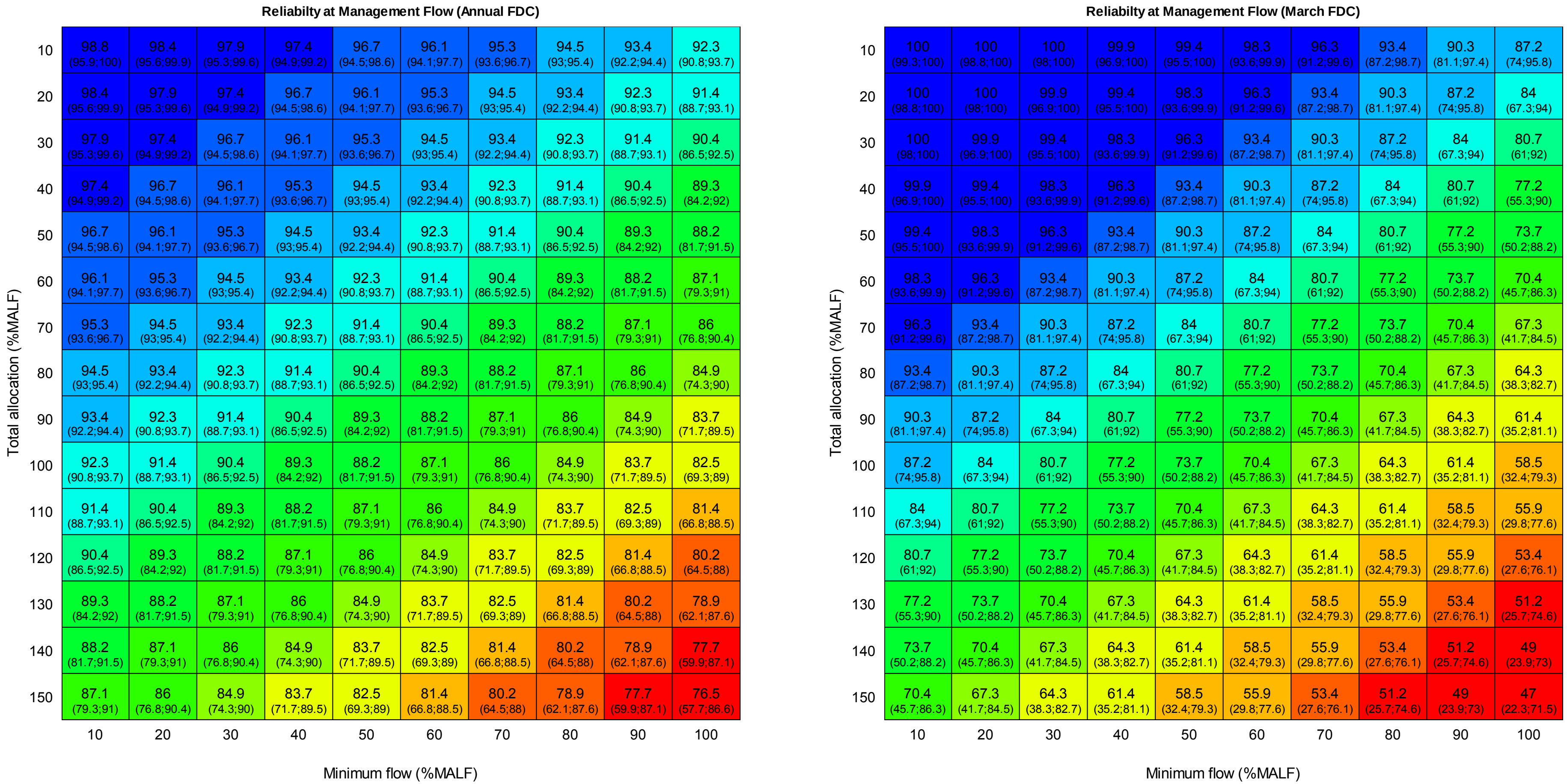
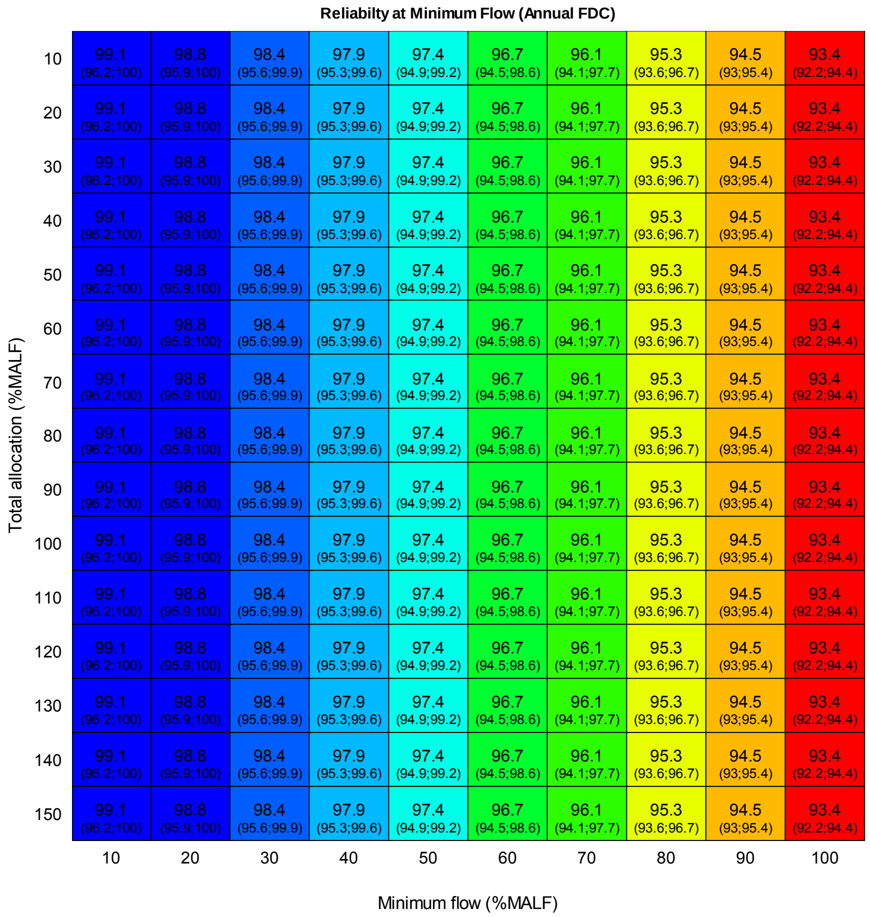


Figure 12: Auckland region decision space diagrams for reliability at management flow. The median value for all locations in the region is presented, with the 10th and 90th percentiles included in brackets. Left: Annual flow duration curves; Right: March flow duration curves. Colours reflect the gradient in median values.



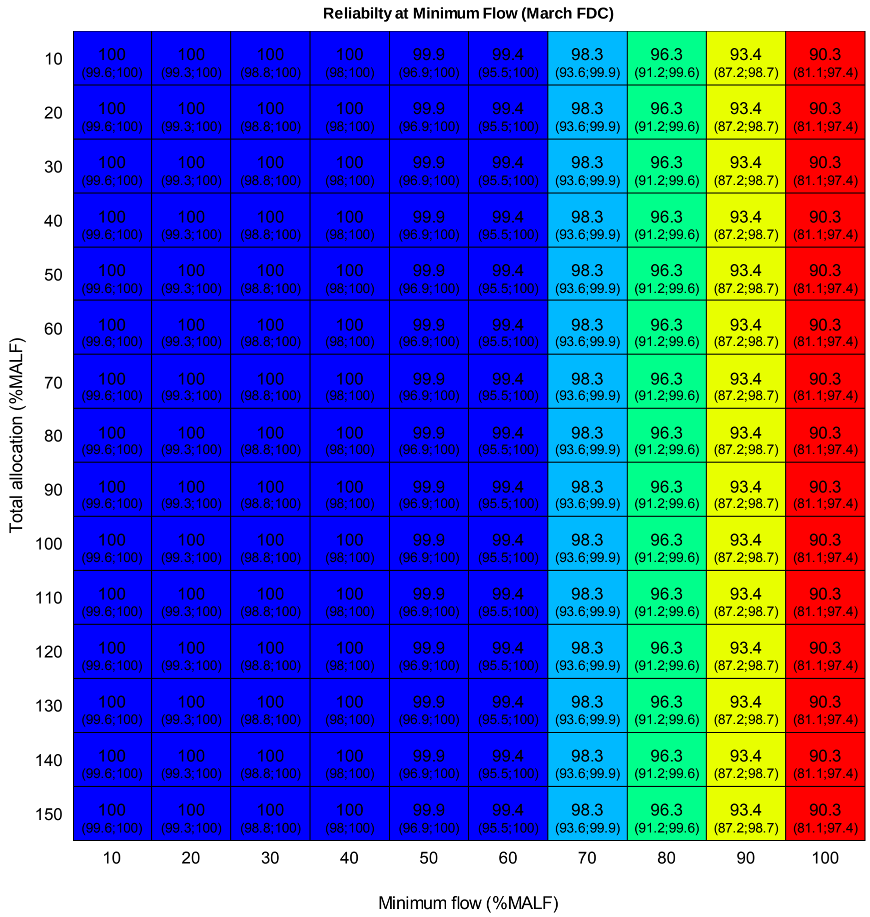
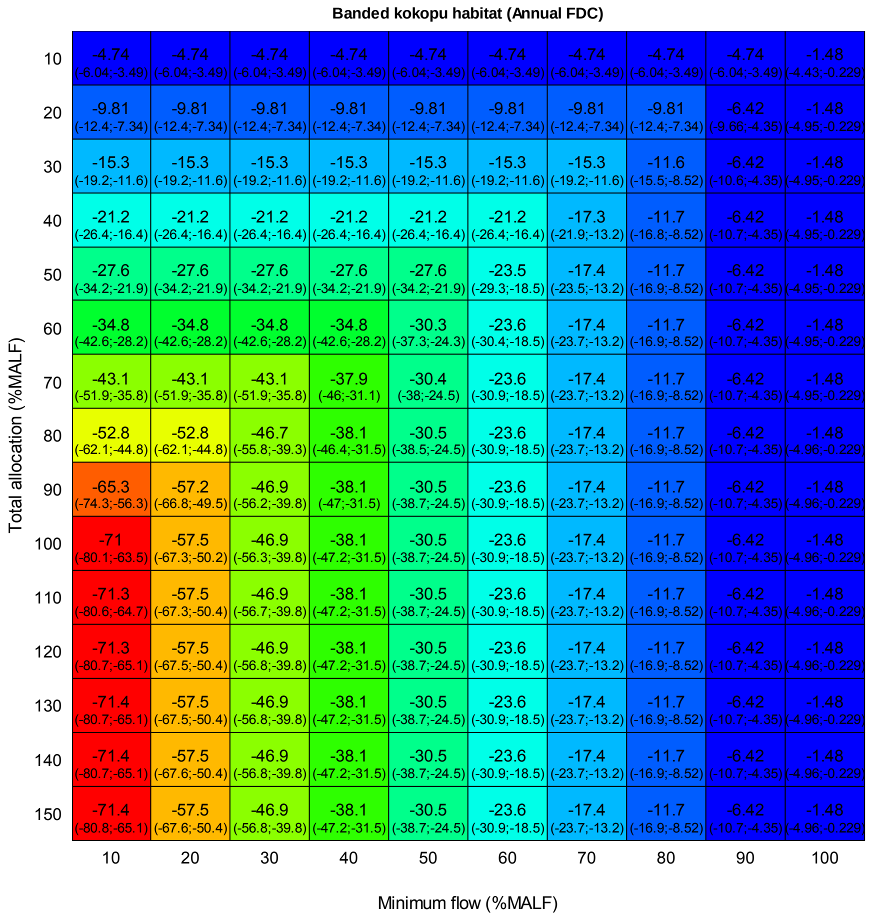


Figure 13: Auckland region decision space diagrams for reliability at minimum flow. The median value for all locations in the region is presented, with the 10th and 90th percentiles included in brackets. Left: Annual flow duration curves; Right: March flow duration curves.



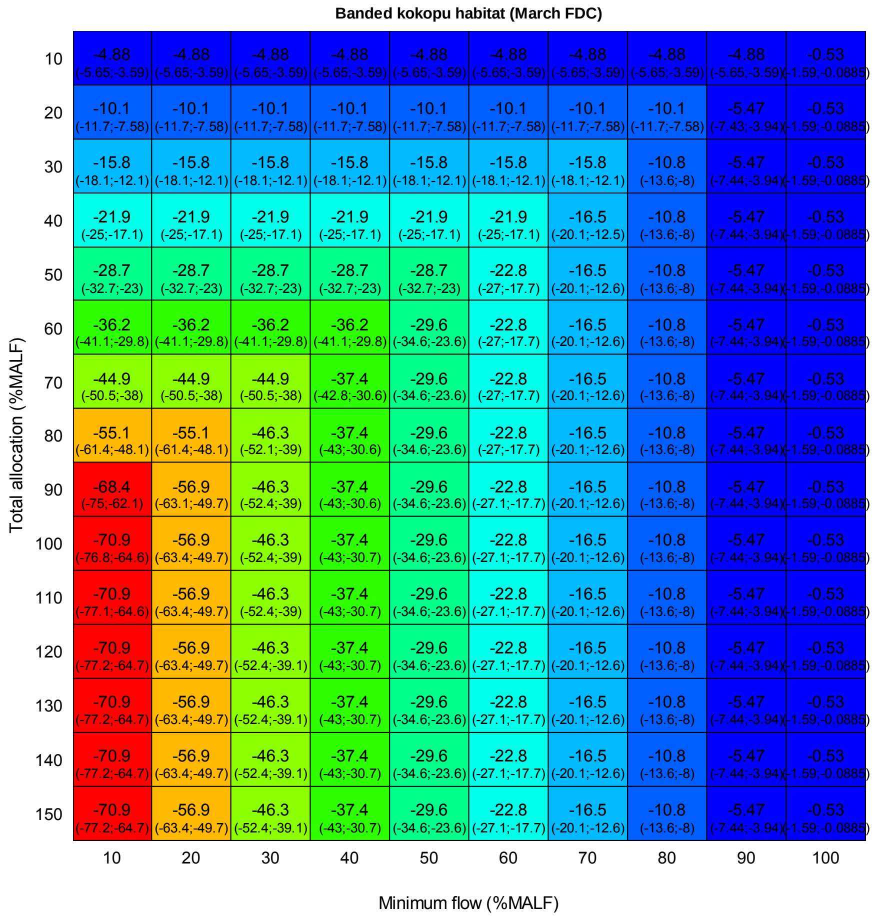
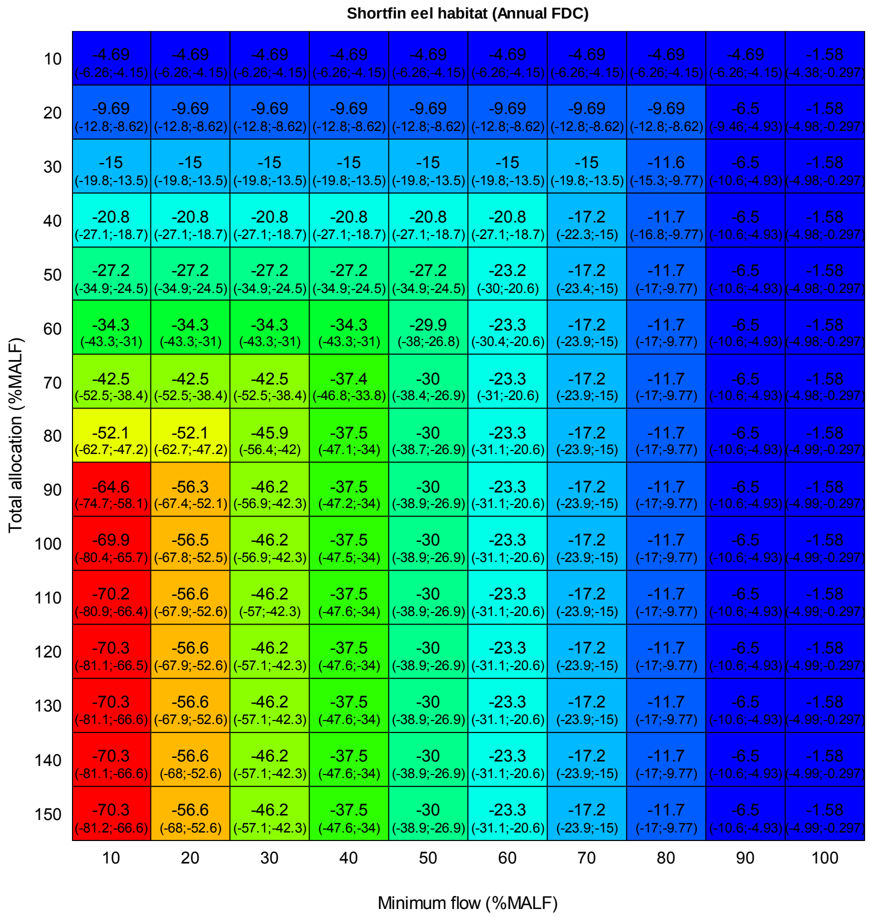


Figure 14: Auckland region decision space diagrams for change in juvenile banded kokopu habitat (% of habitat available at MALF). The median value for all locations in the region is presented, with the 10th and 90th percentiles included in brackets. Left: Annual flow duration curves; Right: March flow duration curves.



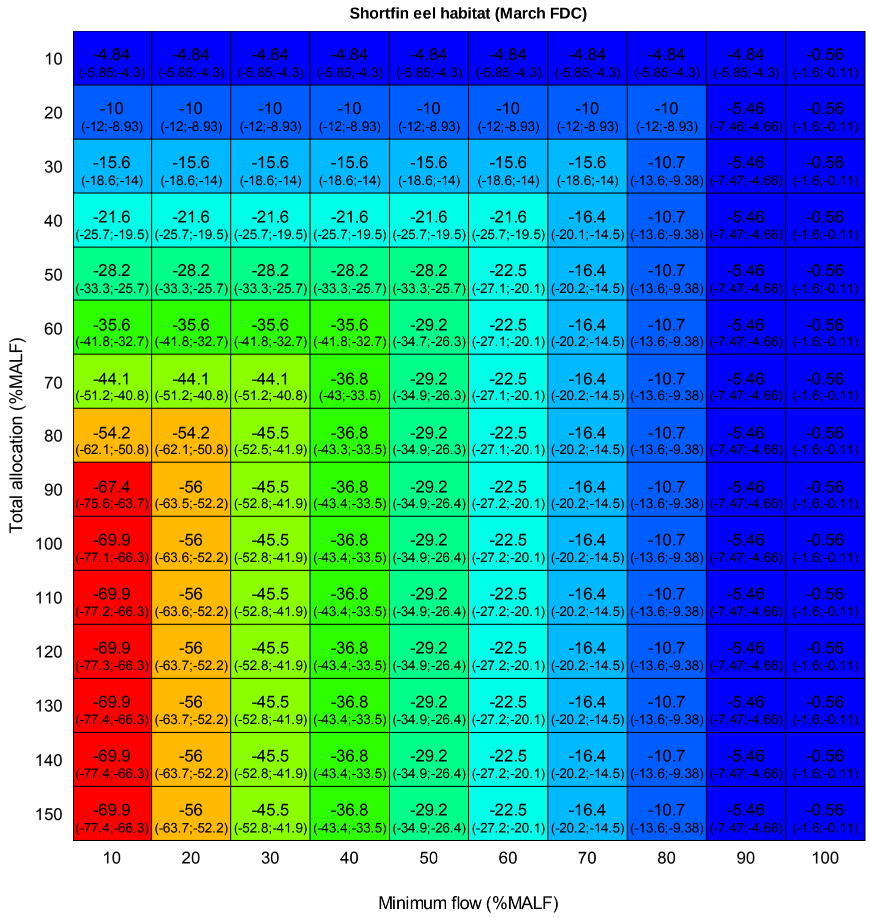
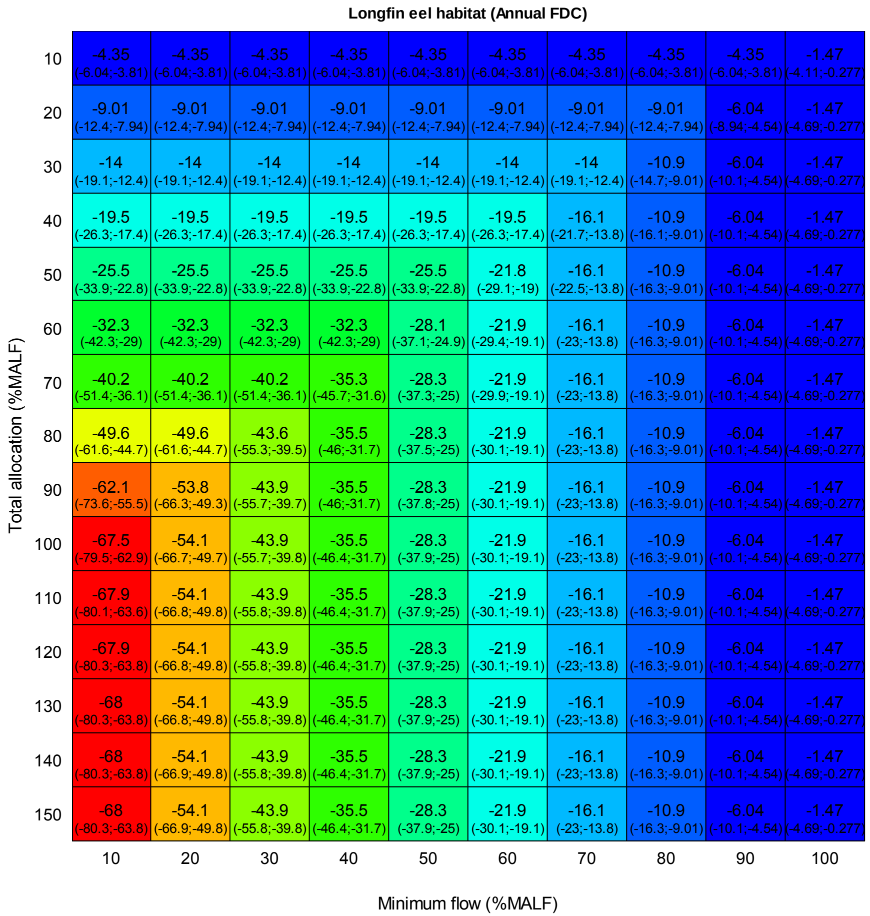


Figure 15: Auckland region decision space diagrams for change in shortfin eel (<30 cm) habitat (% of habitat available at MALF). The median value for all locations in the region is presented, with the 10th and 90th percentiles included in brackets. Left: Annual flow duration curves; Right: March flow duration curves.



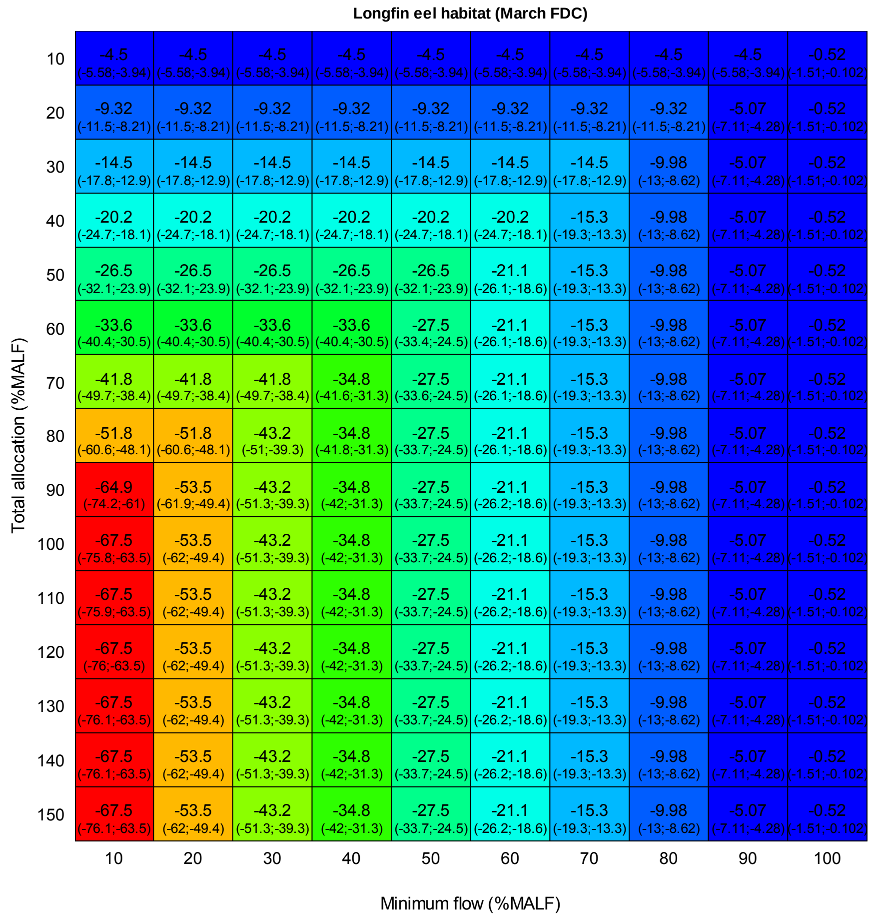
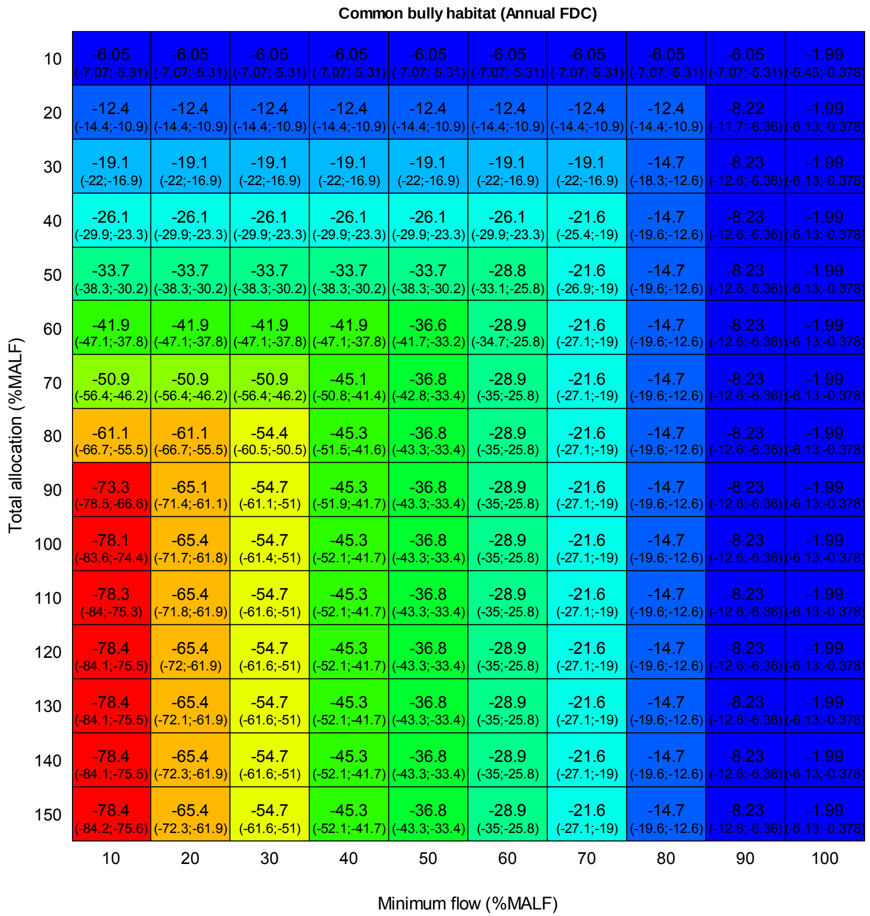


Figure 16: Auckland region decision space diagrams for change in longfin eel (<30 cm) habitat (% of habitat available at MALF). The median value for all locations in the region is presented, with the 10th and 90th percentiles included in brackets. Left: Annual flow duration curves; Right: March flow duration curves.



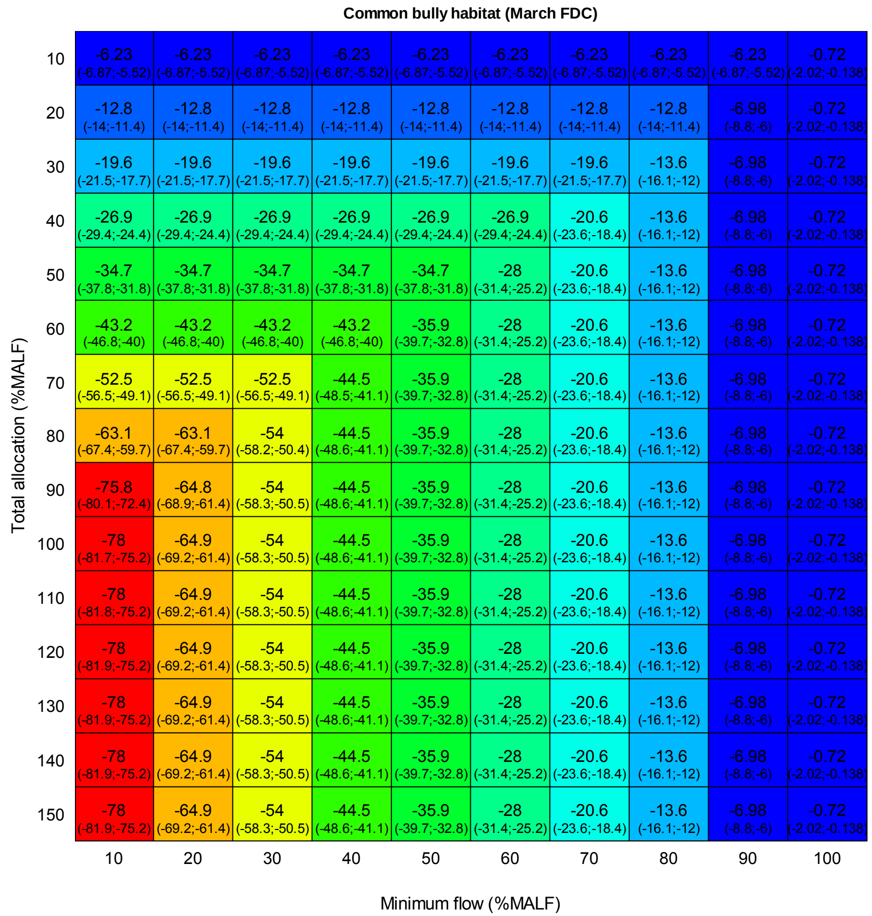


Figure 17: Auckland region decision space diagrams for change in common bully habitat (% of habitat available at MALF). The median value for all locations in the region is presented, with the 10th and 90th percentiles included in brackets. Left: Annual flow duration curves; Right: March flow duration curves.

5.0 Discussion

5.1 Consequences of proposed NES default limits

Table 2 summarises the consequences of applying the proposed NES rules for small rivers (minimum flow of 90 per cent of MALF and total allocation of 30 per cent of MALF) to all locations in the Auckland region. On average across the Auckland region the proposed NES default limits offer relatively good reliability of supply based on the annual FDC, with median reliability at the management flow of 91.4 per cent and 90 per cent of locations having a reliability of at least 88.7 per cent (Table 2). This represents the average amount of time that the full volume of the total allocation would be available for use in an average year. The results based on the March FDC give an indication of the level of reliability during average low flow conditions in summer. As expected, reliability during the summer period would be lower than the overall annual average, with a median reliability of 84.0 per cent and the tenth percentile value being significantly lower at 67.3 per cent (Table 2). Median reliability at the minimum flow is on average 94.5 per cent annually and 93.4 per cent in March (Table 2). This means that abstractions would be fully restricted on average for 5.5 per cent of the year and 6.6 per cent of the time in March.

With respect to the consequences for instream physical habitat for the four indicator species, the proposed NES default limits result in a fairly small loss of habitat relative to that available at MALF. For banded kokopu, shortfin eels and longfin eels, 90 per cent of locations have a reduction in habitat relative to MALF of less than 11 per cent. For common bullies, the loss of habitat is up to 12.6 per cent for 90 per cent of locations (Table 2).

Table 2: Summary of the consequences of applying the proposed NES default limits to the Auckland region. The median and the 10th and 90th percentile values (in parentheses) are shown for each consequence for all locations. Negative values indicate a decrease in habitat from that available at MALF.

FDC	Reliability at management flow (%)	Reliability at minimum flow (%)	Banded kokopu	Shortfin eel	Longfin eel	Common bully
Annual	91.4 (88.7; 93.1)	94.5 (93.0; 95.4)	-6.42 (-10.6; -4.35)	-6.50 (-10.6; -4.93)	-6.04 (-10.1; -4.54)	-8.23 (-12.6; -6.36)
March	84.0 (67.3; 94.0)	93.4 (87.2; 98.7)	-5.47 (-7.44; -3.94)	-5.46 (-7.47; -4.66)	-5.07 (-7.11; -4.28)	-6.98 (-8.8; -6.0)

The proposed NES default limits are only one of a range of options available to river managers for managing water resource use. The challenge for water resource managers is to evaluate the relative advantages and disadvantages of different allocation scenarios so that an informed decision can be made on the most appropriate limit combinations. The

EFSAP decision space diagrams can facilitate this process by summarising the regional scale consequences of particular water resource use limit combinations. The following section describes how the EFSAP decision space diagrams can be used to support the decision making process.

5.2 Use of decision space figures for limit setting

Definition of water resource use limits involves a trade-off between different instream and out-of-channel uses of water. EFSAP is used to evaluate the consequences of a range of scenarios for water allocation limits, thereby enabling water resource managers to make a more informed and transparent choice. The following guidance is provided to explain how to use the EFSAP modelling outputs presented in this report for choosing limits from amongst the different scenarios.

The scenarios assessed using EFSAP, and their associated consequences for reliability and instream habitat, are presented as 'decision space' diagrams (e.g., Figures 12 to 17). The decision space summarises the consequences for each scenario, i.e., combination of minimum flow and allocation limit. For each scenario, the median, 10th and 90th percentiles of the consequences for the values (i.e., the reliability or change in habitat) are presented (Figure 18). These percentiles summarise the consequences of that combination of limits for all locations in a management unit. The median is used to represent the 'average' consequence for that value across all locations in a management unit. In some locations the consequence will be worse, and in some locations better. The difference between the 10th and 90th percentiles gives an indication of how variable the consequences for a value are across all locations within the management unit. The greater the difference between these figures, the larger the variation in consequences between locations, the smaller the difference between the figures, the more uniform and thus equitable the consequences are between locations in the management unit. Ninety percent of locations in the management unit will have a consequence which is at least as good as the 10th percentile.

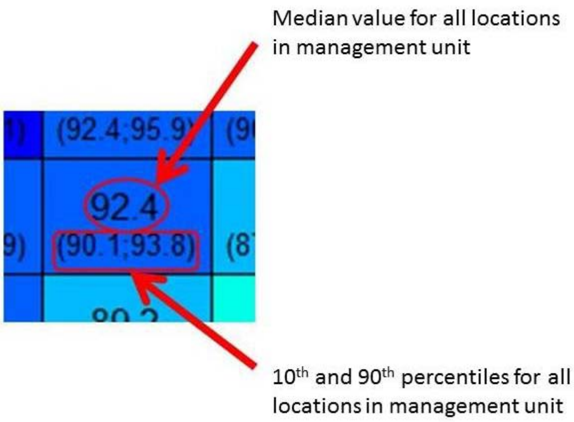


Figure 18: Interpreting the decision space diagram summary statistics.

The first task in determining appropriate water resource use limits should be the determination of objectives for each value. Ideally objectives should be clear (i.e., defined levels of protection for environmental values and availability and reliability of water for out-of-channel uses), transparent (i.e., stakeholders can understand why they have been selected as objectives) and measureable (i.e., it is possible to measure whether the objectives are being met). Once objectives have been set, the decision space diagrams can be used to determine which combination of limits satisfies the objectives (e.g., Figure 19).

For illustrative purposes, we arbitrarily set objectives of a median reliability at management flow of ≥90 per cent, median reliability at minimum flow of ≥95 per cent and a median loss of habitat ≤15 per cent. Having defined these objectives, the combinations of minimum flow and total allocation which satisfy the objective for each value can be determined from the decision space diagrams. In Figure 19, this is represented by the areas enclosed by thick black lines on the decision space for each of the three values (top panels and bottom-left panel). For this example, the black lines are defined by whether the median value in each square meets the objective threshold, but potentially other percentiles could be used. Once the subset of limits that satisfy the objectives for each individual value have been defined, they are combined to find the set of limits which meet all objectives. This is illustrated in the bottom-right panel of Figure 19, where the grey shaded area represents the intersection of the three objectives and therefore the combinations of minimum flow and total allocation that satisfy all three objectives. Further decision space diagrams for different values, or the same values at different times of the year, can be added to the analysis, and their associated objectives used to further constrain the possible range of suitable limits.

In this example, the defined objectives for all three values result in a combination of limit options that overlap (Figure 19). Water resource managers therefore have the choice of defining limits that satisfy all objectives. However, even within this more constrained set of options for limits, value judgments are required to define the final choice of limits. For example, options could include maximising environmental protection (minimum flow of 80 per cent of MALF and total allocation of 10 per cent of MALF), maximising reliability (minimum flow of 10 per cent of MALF and total allocation of 10 per cent of MALF), or maximising total allocation (minimum flow of 80 per cent of MALF and total allocation of 50 per cent of MALF). The decision will vary based on the relative importance of the different instream and out-of-stream values assessed, and may vary between management units (e.g., in the Auckland region objectives in NSMAs may be different to other streams). In making the decision on the most appropriate combination of limits, it should also be remembered that values additional to those evaluated by EFSAP may be important and may therefore help to guide the decision making process. For example, a minimum flow of 10 per cent of MALF (which satisfies all three of the objectives here), combined with a modest allocation (e.g., 20 per cent of MALF) may not be considered acceptable due to the excessive departure from the natural range of variability that may typify natural character.

It is possible that under alternative objectives a situation may arise whereby no combination of limits would satisfy all objectives concurrently. In this situation, a compromise has to be found between the different values until an acceptable combination of limits can be agreed upon. The decision space diagrams can assist in this trade-off process by illustrating to stakeholders and resource managers how limits interact with each other and the relative consequences of alternative management decisions. This makes the process of limit setting more transparent and accountable.

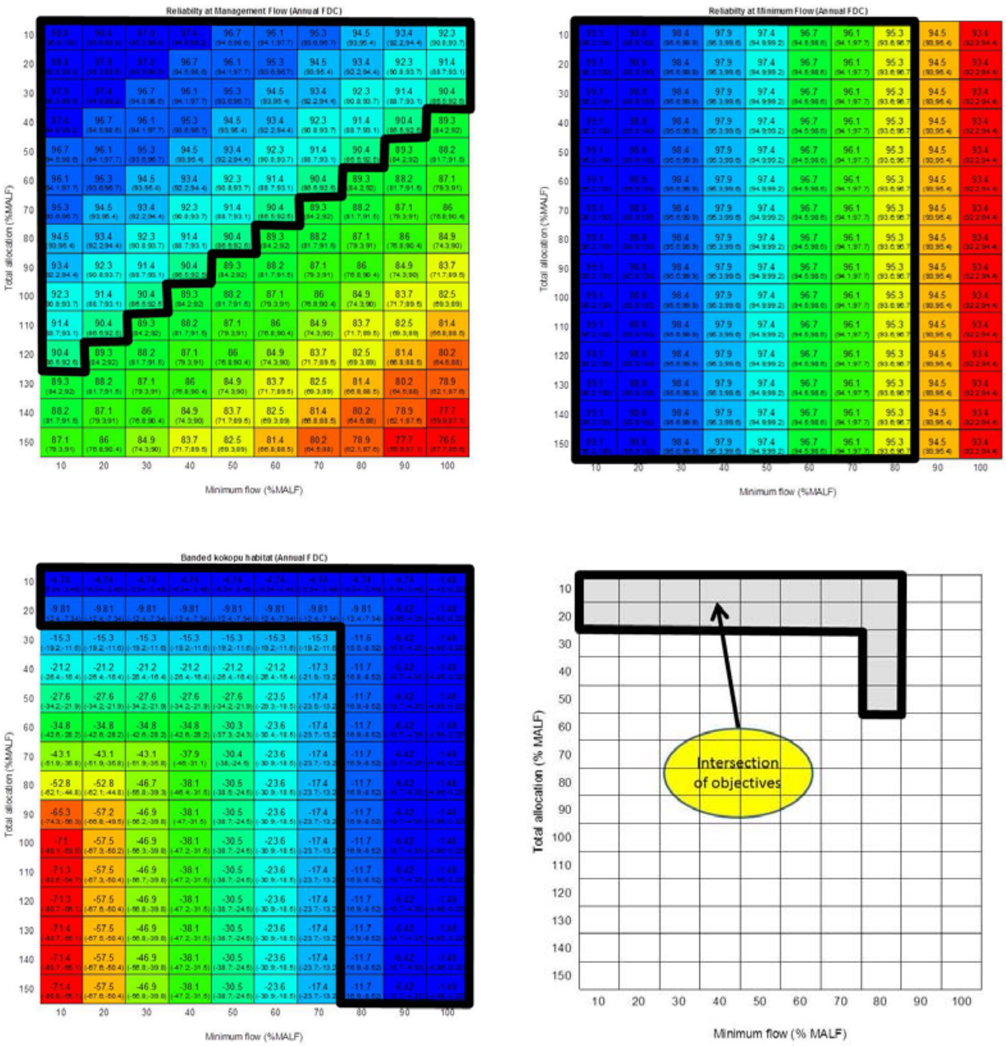


Figure 19: Example of evaluating objectives for multiple values and determining range of limits that satisfy all objectives, as described in the text above.

5.3 Limitations

Snelder et al. (2011) identified six limitations associated with the methodology that has been used for this study, which should be acknowledged and taken into consideration when interpreting and applying the results presented above. First, the concept of flow variability was not explicitly considered in the analysis. Flow variability is increasingly acknowledged as being critical for ecosystem health and therefore should be considered in setting environmental flows (Poff et al. 1997, Poff et al. 2010). The EFSAP methodology is primarily designed to evaluate the effects of run-of-river abstractions, where total allocation is low relative to the mean flow. This type of water use primarily affects the flow regime in terms of the magnitude and duration of low flows, but tends to have relatively little effect on medium to high flows. Where more intensive water resource development that significantly alters the flow regime has, or is expected to occur, e.g., damming or water diversion, a more detailed, site specific assessment would be required that explicitly considered the effects on flow variability.

A second limitation is that FDCs provide no information regarding the temporal sequencing of flows. It is therefore not possible to determine whether periods of restriction or time at minimum flows occur consecutively or scattered through time. This is partially alleviated by providing results based on monthly FDCs. In this study, for example, we have reported results for the March FDC as being representative of the most resource restrictive period. Analysis of natural flow time series would be required if more detail on the timing and temporal sequencing of restrictions was needed.

Another limitation of this study was that the uncertainties associated with any estimate were not evaluated. The analysis was dependent on estimates of MALF and FDCs. Uncertainties around the estimation of these parameters can be large, especially around the low flows that this analysis focussed on (Booker & Woods 2012). In addition, the at-station hydraulic geometry and generalized physical habitat model uncertainties were propagated through the various analyses. This means that the observed patterns are probably indicative of the relative differences at a regional scale, but that the uncertainties for individual segments could be large. Future work by NIWA will aim to quantify the total uncertainty of the predictions, as well as decrease the uncertainties associated with individual models.

A fourth limitation to our approach is that the complexity of flow management was simplified. We treated each stream segment as independent and made an assessment of the consequences on physical habitat and reliability as though the minimum flow was observed at that segment and that allocation, and therefore total abstraction, occurred in its upstream catchment. In reality, abstractions are distributed unevenly in space and the consequences accumulate down the river network in a non-uniform manner. This means that consequences for habitat retention and reliability across the network can be more variable than shown in our analysis. In addition, it was assumed that all water abstractions were direct and did not include groundwater abstractions that may affect river flows in a different and less direct way.

A fifth limitation concerns the assumption that the quantity of physical habitat is an appropriate indicator of ecosystem protection during low flow periods. We used the proportional change in the availability of physical habitat at MALF to compare the consequences for instream values. This assumes that ecosystems are naturally stressed at low flows, but this may not always be the case. In some locations, for some species, other factors such as water quality, temperature, and migration pathways may be more important controls. Other flow dependent values such as recreation or cultural values may also be more significant. Despite these limitations, the use of changes in physical habitat to evaluate the consequences of flow change is well established in New Zealand and worldwide (Beca 2008, MfE 1998).

A final limitation is that this analysis was restricted to only a selected number of indicator taxa. These taxa were selected based on known and predicted distributions (i.e. NZFFD; Leathwick et al. 2008) and their conservation status (Allibone et al. 2010) to maximise their relevance, but the choice of taxa was still restricted to those for which generalised habitat models are available (Appendix A, Table 3). A full analysis for environmental flow setting would require inclusion of multiple species and life stages, and an acknowledgement of the interdependence between taxa and life stages. An advantage of the EFSAP methodology over other habitat modelling approaches is that it facilitates better integration of multiple taxa into the limit setting process through the option to overlap multiple decision spaces. Ideally, however, water resource use limits would also be based on linking physical habitat availability and quality to population dynamics (e.g. Capra et al. 2003).

6.0 Conclusions

The NPS for Freshwater Management (MfE 2011) requires that regional councils define environmental flow limits that include both minimum flows and total allocation limits. Where demand for water resources is high, detailed, site specific assessments will be required to define limits. However, there is also a need to define limits for less intensively developed catchments in a cost effective and transparent way.

The EFSAP methodology provides a new approach for water resource managers to evaluate the consequences of setting different water resource use limits across all parts of a catchment or region including those for which detailed information is not available. The integrated use of scientific tools allows concurrent evaluation of consequences for both instream habitat and reliability of supply for out-of-channel water uses. It also accounts for the interaction between minimum flow and total allocation limits. By modelling a range of scenarios it also allows resource managers to more effectively communicate to stakeholders the varying consequences of different water resource limits.

In combination with clearly defined and measurable objectives for managing water resource use, the EFSAP outputs facilitate definition of transparent and justifiable water resource use limits. However, water resource managers will still be required to make value judgments and balance the needs of competing values. They will also need to consider values that are not evaluated by EFSAP such as cultural, natural character and recreation values. Consequently, the decision making will remain an iterative process involving multiple stakeholders with competing demands and values.

7.0 Acknowledgements

We would like to acknowledge the assistance of Auckland Council staff in identifying indicator species and supplying spatial data. The development of EFSAP was funded by the Ministry for Science and Innovation.

8.0 References

Acreman, M.; Dunbar, M.; Hannaford, J.; Mountford, O.; Wood, P.; Holmes, N.; Cowx, I.; Noble, R.; Extence, C.; Aldrick, J. (2008). Developing environmental standards for abstractions from UK rivers to implement the EU Water Framework Directive/Développement de standards environnementaux sur les prélèvements d'eau en rivière au Royaume Uni pour la mise en œuvre de la directive cadre sur l'eau de l'Union Européenne. *Hydrological Sciences Journal 53(6)*: 1105–1120.

Allibone, R.; David, B.; Hitchmough, R.; Jellyman, D.J.; Ling, N.; Ravenscroft, P.; Waters, J. (2010). Conservation status of New Zealand freshwater fish, 2009. *New Zealand Journal of Marine & Freshwater Research 44(4)*: 271–287.

Beca (2008). Draft guidelines for the selection of methods to determine ecological flow and water levels. Report prepared by Beca Infrastructure Ltd. 145 p.

Booker, D.J. (2010). Predicting wetted width in any river at any discharge. *Earth Surface Processes and Landforms 35*: 828–841.

Booker, D.J.; Dunbar, M.; Ibbotson, A.T. (2004). Predicting juvenile salmonid drift-feeding habitat quality using a three-dimensional hydraulic-bioenergetic model. *Ecological Modelling 177*: 157–177.

Booker, D.J.; Woods, R. (2012). Hydrological estimates for Auckland. Prepared by NIWA for Auckland Council. Auckland Council technical report TR2012/042.

Capra, H.; Sabaton, C.; Gouraud, V.; Souchon, Y.; Lim, P. (2003). A population dynamics model and habitat simulation as a tool to predict brown trout demography in natural and bypassed stream reaches. *River Research & Applications 19(56)*: 551–568.

Clausen, B.; Jowett, I.G.; Biggs, B.J.F.; Moeslund, B. (2004). Stream ecology and flow management. In: Tallaksen, L.M.; Van Lanen, H.A.J. (eds). *Developments in water science 48*, pp. 411–453. Elsevier, Amsterdam.

Dunbar, M.J.; Acreman, M.C. (2001). Applied hydro-ecological science for the twenty-first century. *In:* Acreman, M.C. (ed.). Hydro-ecology: Linking hydrology and aquatic ecology - Proceedings of Birmingham workshop, July 1999, pp. IAHS, Birmingham.

Jowett, I.G. (1996). RHYHABSIM river hydraulics and habitat simulation computer manual. Hamilton, NIWA. p.

Jowett, I.G. (1998). Hydraulic geometry of New Zealand rivers and its use as a preliminary method of habtiat assessment. Regulated Rivers: *Research & Management 14*: 451–466.

Jowett, I.G.; Biggs, B.J.F. (2006). Flow regime requirements and the biological effectiveness of habitat-based minimum flow assessments for six rivers. *International Journal of River Basin Management 4(3)*: 179–189. <http://dx.doi.org/10.1080/15715124.2006.9635287>

Jowett, I.G.; Hayes, J.W.; Duncan, M.J. (2008). A guide to instream habitat survey methods and analysis. N*IWA Science and Technology Series No. 54*. 121 p.

Lamouroux, N. (2008). Hydraulic geometry of stream reaches and ecological implications. *In:* Habersack, H.; Piégay, H.; Rinaldi, M. (eds). Gravel Bed Rivers 6: From process understanding to the restoration of mountain rivers, pp. 661–675. Developments in Earth Surface Processes. Elsevier, Amsterdam.

Lamouroux, N.; Capra, H. (2002). Simple predicitons of instream habitat model outputs for target fish populations. *Freshwater Biology 47*: 1543–1556.

Lamouroux, N.; Jowett, I.G. (2005). Generalized instream habitat models. *Canadian Journal of Fisheries and Aquatic Sciences 62(1)*: 7–14.

Lamouroux, N.; Mérigoux, S.; Capra, H.; Dolédec, S.; Jowett, I.G.; Statzner, B. (2010). The generality of abundance-environment relationships in microhabitats: A comment on Lancaster and Downes (2009). *River Research & Applications 26(7)*: 915–920. <http://dx.doi.org/10.1002/rra.1366>

Lamouroux, N.; Souchon, Y. (2002). Simple predicitons of instream habitat model outputs for fish habitat guilds in large streams. *Freshwater Biology 47*: 1531–1542.

Lancaster, J.; Downes, B.J. (2010). Linking the hydraulic world of individual organisms to ecological processes: Putting ecology into ecohydraulics. *River Research and Applications 26(4)*: 385–403. <http://dx.doi.org/10.1002/rra.1274>

Leathwick, J.R.; Julian, K.; Elith, J.; Rowe, D.K. (2008). Predicting the distributions of freshwater fish species for all New Zealand's rivers and streams. *NIWA Client Report No. HAM2008-005*. 56 p.

Mathur, D.; Bason, W.; Purdy, E.; Silver, C. (1985). A critique of the instream flow incremental methodology. *Canadian Journal of Fisheries and Aquatic Science 42*: 825–831.

MfE (1998). Flow guidelines for instream values - Part A. 146 p.

MfE (2008). Proposed National Environmenal Standard on ecological flows and water levels. *Ministry for the Environment Discussion Document No. ME 868*. 61 p.

MfE (2011). National Policy Statement for Freshwater Management 2011. 12 p.

Nilsson, C.; Renöfält, B.M. (2008). Linking flow regime and water quality in rivers: a challenge to adaptive catchment management. *Ecology and Society 13(2)*: 18.

Orth, D.J. (1987). Ecological considerations in the development and application of instream flow-habitat models. *Regulated Rivers: Research and Management 1*: 171–181.

Poff, N.L.; Allan, J.D.; Bain, M.B.; Karr, J.R.; Prestegaard, K.L.; Richter, B.D.; Sparks, R.E.; Stromberg, J.C. (1997). The natural flow regime: a paradigm for river conservation and restoration. *BioScience 47*: 769–784.

Poff, N.L.; Richter, B.D.; Arthington, A.H.; Bunn, S.E.; Naiman, R.J.; Kendy, E.; Acreman, M.; Apse, C.; Bledsoe, B.P.; Freeman, M.C.; Henriksen, J.; Jacobson, R.B.; Kennen, J.G.; Merritt, D.M.; O'Keeffe, J.H.; Olden, J.D.; Rogers, K.; Tharme, R.E.; Warner, A. (2010). The ecological limits of hydrologic alteration (ELOHA): a new framework for developing regional environmental flow standards. *Freshwater Biology 55(1)*: 147–170. <http://dx.doi.org/10.1111/j.1365-2427.2009.02204.x>

Reiser, D.W.; Wesche, T.A.; Estes, C. (1989). Status of instream flow legislation and practices in North America. *Fisheries Management and Ecology 14*: 22–29.

Richter, B.D.; Warner, A.T.; Meyer, J.L.; Lutz, K. (2006). A collaborative and adaptive process for developing environmental flow recommendations. *River Research and Applications 22*: 297–318.

Snelder, T.; Biggs, B.J.F. (2002). Multi-scale river environment classification for water resources management. *Journal of the American Water Resources Association 38(5)*: 1225–1239. <http://dx.doi.org/10.1111/j.1752-1688.2002.tb04344.x>

Snelder, T.; Booker, D.; Lamouroux, N. (2011). A method to assess and define environmental flow rules for large jurisdictional regions. *Journal of the American Water Resources Association 47(4)*: 828–840.

Snelder, T.; Hughey, K.F.D. (2005). On the use of an ecological classification to improve water resource planning in New Zealand. *Environmental Management 36(5)*: 741–756.

Snelder, T.; Weatherhead, M.; Biggs, B.J.F. (2004). Nutrient concentration criteria and characterization of patterns in trophic state for rivers in heterogeneous landscapes. *Journal of the American Water Resources Association 40(1)*: 1–13.

Vogel, R.M.; Fennessey, N.M. (1995). Flow duration curves II: A review of applications in water resources planning. *Journal of the American Water Resources Association 31(6)*: 1029–1039. <http://dx.doi.org/10.1111/j.1752-1688.1995.tb03419.x>

Appendix A

Table 3: Species for which generalised habitat models are available in New Zealand. The model parameters *c* and *k* are displayed and optimum discharge per unit width provides an indication of relative flow demand (Source: Jowett et al. 2008).

Species	*c*	*k*	Optimum discharge per unit width (m^2s^{-1})
Inanga	0.19	19.74	0.01
Shortjaw kokopu	0.19	16.35	0.01
Upland bully	0.11	8.63	0.01
Cran's bully	0.09	6.84	0.01
Banded kokopu (juvenile)	0.19	13.3	0.01
Canterbury galaxias	0.03	2.29	0.01
Roundhead galaxias	0.31	10.64	0.03
Flathead galaxias	0.28	9.11	0.03
Longfin eel (<30 cm)	0.07	2.07	0.03
Lowland longjaw galaxias	0.33	9.35	0.04
Redfin bully	0.26	7.39	0.04
Shortfin eel (<30 cm)	0.13	2.32	0.05
Common bully	0.39	6.51	0.06
Brown trout fry	0.86	10.21	0.08
Brown trout yearling	0.40	4.18	0.09
Nesameletus	0.26	2.62	0.10
Brown trout spawning	1.24	9.89	0.13
Bluegill bully	1.01	6.13	0.16
Rainbow trout spawning	1.49	8.78	0.17
Deleatidium	0.33	1.92	0.17
Torrentfish	0.88	4.05	0.22
Brown trout adult	1.17	4.35	0.27
Food producing habitat	1.19	4.25	0.28
Rainbow trout feeding (30-40 cm)	0.93	2.89	0.32
Coloburiscus humeralis	1.35	4.17	0.32
Aoteapsyche	1.44	3.17	0.45
Zelandoperla	1.71	3.40	0.50

Find out more: phone 09 301 0101
email research@aucklandcouncil.govt.nz
or visit www.aucklandcouncil.govt.nz

Auckland Council
Te Kaunihera o Tāmaki Makaurau