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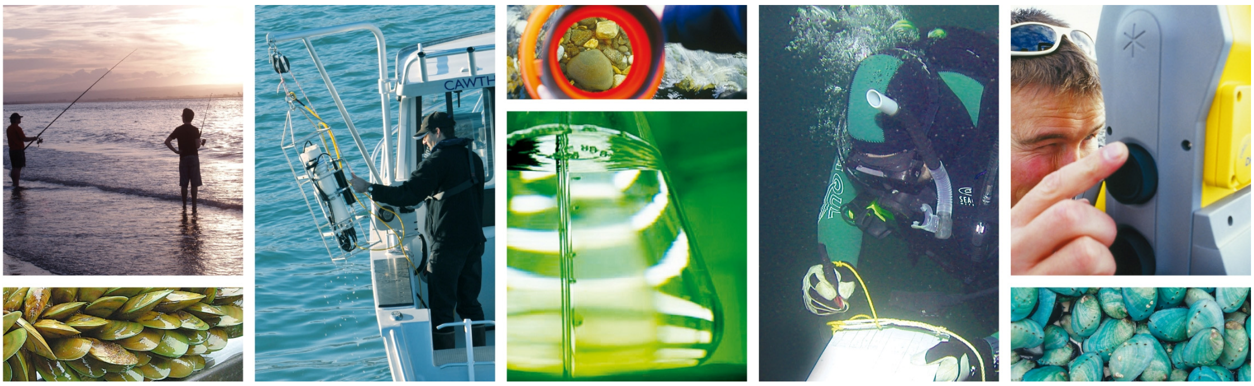
Aquatic Ecology: Mitigation and Management Options Associated with Water Storage in the Proposed Lee Reservoir



RESEARCH BASED SOLUTIONS

Aquatic Ecology: Mitigation and Management Options Associated with Water Storage in the Proposed Lee Reservoir

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Prepared for

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on behalf of

Waimea Water Augmentation Committee

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EXECUTIVE SUMMARY

This report presents the results of aquatic ecology investigations completed as part of the Phase 2 feasibility study for a potential dam on the Lee River in Tasman District. The proposed dam is part of a water augmentation scheme to capture water for storage, and release that water back into the Waimea River system during periods of high water demand and/or low natural flows. In this report we provide information on existing water quality and invertebrate communities in the Lee River at the proposed dam site, examine the potential for stratification in the proposed reservoir and implications of different water outlet levels, recommend appropriate minimum and flushing flows for the Lee River downstream of the proposed dam, comment on mitigation of fish passage issues associated with the dam, and address potential issues with reservoir management that may affect fishery values in the reservoir.

Existing water quality data from the Lee River at the proposed dam site was indicative of the undeveloped nature of catchment upstream and well within national guidelines for protection of river ecosystem health. The values for most water quality parameters at the proposed dam site were very similar to those further downstream at the Tasman District Council's monitoring site on the Lee River at Meads Bridge. The only exceptions were concentrations of nitrate-nitrogen and total nitrogen which were elevated at the Meads Bridge site, perhaps reflecting the increased proportion of developed land in the lower part of the Lee Catchment. Stream invertebrate communities at the proposed dam site and at Meads Bridge were equivalent and generally indicative of very good stream health. No rare species of invertebrates were recorded at either site.

Most deep lakes will stratify during summer with a layer of warm surface water 'floating' above a cooler, more dense, bottom layer. During periods of stratification, oxygen demand in the bottom of a lake can lead to reductions in dissolved oxygen concentration in the bottom waters. In severe cases this can lead to anoxic conditions. A computer model (DYRESM) was used to simulate water temperatures throughout the water column in the proposed Lee Reservoir, which would have a depth up to approximately 46 m. The model predicted cool temperatures throughout the water column in winter indicating a well mixed water column. Stratification was predicted to occur in late spring, summer and autumn with the thermocline (boundary between surface and bottom waters) approximately 10 m below the reservoir surface during wet years and much deeper (30+ m) in dry years. The level selected for the outlet is predicted to have a large influence on where the stratification layer develops. We recommend that two outlets are incorporated into the scheme design. One outlet should be near the bottom of the reservoir to be used during dry years and during floods to flush any poor quality bottom water from the reservoir. A second outlet should be at approximately 185 m Relative Level (RL) to be used under most conditions to release good quality surface water from the reservoir during most years.

Predictions of the trophic status of lakes can be made based on relationships between nutrient concentrations and observed concentrations of algal biomass and water clarity. Assuming that the current nutrient loading in the Lee River at the dam site will remain the same after dam construction, the reservoir is predicted to be oligotrophic with low concentrations (<2 mg/m^3) of phytoplankton biomass and relatively high water clarity (7-15 m). These predictions match observations from the

neighbouring Maitai Reservoir, which experiences similar climatic conditions and has similar land-use in the catchment upstream.

During the Phase 1 pre-feasibility study a flow equal to the existing mean annual low flow (MALF) was suggested as the minimum flow for the Lee River below the dam. During the current study, habitat modelling was conducted to provide a more robust indication of instream habitat requirements at the proposed dam site. Habitat modelling was conducted with the computer programme RHYHABSIM. The natural 7-day MALF (0.51 m^3/s) is proposed as the environmental benchmark minimum flow for the Lee River below the proposed dam. A minimum flow of 0.32 m^3/s would retain 70% of the yearling to adult brown trout habitat available at the natural MALF. An intermediate option is a minimum flow of 0.38 m^3/s, which would retain 80% of the habitat available at the natural MALF for yearling to adult brown trout. The flow requirements for trout spawning are slightly higher than those for yearling and adult trout. For this reason a higher minimum flow (0.41 m^3/s for 80% habitat retention compared with the MALF; 0.35 m^3/s for 70% habitat retention compared with the MALF) may be warranted during the winter spawning and egg incubation season (May to November). Minimum flows based on habitat retention for native fish were consistently lower than for brown trout. This supports the contention that setting a minimum flow to protect trout habitat availability should also accommodate the minimum flow requirements of native fish. The Waimea Water Augmentation Committee has chosen to adopt our recommendation that the natural 7-day MALF (0.51 m^3/s) shall be the environmental benchmark minimum flow for the Lee River immediately below the proposed dam. The committee has also adopted a minimum flow of 1.1 m^3/s for the Waimea River at Appleby (after all water demands have been accounted for). For comparison, the existing summer minimum flow stipulated in the proposed Tasman Resource Management Plan is 0.5 m^3/s.

It is possible that hydro-power generation will be incorporated into the scheme design. Synthesised flow regimes downstream of the dam based on without-hydro and base-load hydro scenarios indicate very minor effects on key ecologically relevant flow statistics and are predicted to have a negligible effect on habitat availability. However, any potential effects of hydro-peaking on habitat availability downstream were beyond the scope of this study. Therefore, if hydro-peaking were to be considered during a later hydro-power optimisation study, then the specific effects on habitat would need to be assessed.

Using the RHYHABSIM model it is possible to predict the maximum suspended sediment and maximum bedload particle size that will be mobilised under different flows. This analysis indicated that a flow of about 3 m^3/s would be required to initiate flushing of fine sediment and periphyton. However, flows of 4.5–5.0 m^3/s were predicted to be required before appreciable effects of flushing would occur. Commonly used rules of thumb for periphyton flushing flows of 3 times the median flow, or 6-8 times the baseflow, support the suggestion that a flow of 5 m^3/s would be an appropriate flushing flow. A dam designed with an outlet capable of releasing 5 m^3/s of clean surface water would provide the potential to mitigate any accumulations of nuisance periphyton that occur below the dam.

After weighing up the positive and negative effects of the proposed storage scheme on instream habitat availability, most species (trout, eels, torrentfish, koaro, upland bully) are predicted to benefit from the augmentation scheme as a result of increased flows in the lower Waimea River during dry periods. The main exception is redfin bully, which tend to like slow shallow water, and thus will not benefit from enhanced minimum flows in the lower reaches of the river system. Redfin bullies will also be unlikely to negotiate the fish pass and continue to occupy habitat above the dam.

Given the height of the proposed dam (approximately 52 m to the dam crest) and the relatively low status of the trout fishery in the Lee River, it is considered that mitigation of fish passage issues associated with the dam is only necessary and practical for the strongest of migrants such as elvers and young koaro. Iterative discussions on fish passage have been held between the dam designer (Tonkin & Taylor), Cawthron, Fish & Game New Zealand, and Department of Conservation. Several initial design options were ruled out during this process due to their incompatibility with fish passage. The currently agreed solution (which is incorporated in the proposed dam design) is a nature-like fish passage channel crossing the downstream face of the dam from left to right and flowing out adjacent to the attraction flow provided by the augmentation flow release outlet. In order to ensure continual flow and fish passage through the fish passage channel a piped flow will need to be directed into the fish passage channel at the dam crest. The most difficult issue to deal with at this stage is downstream migration of adult eels. The flip bucket at the bottom of the spillway has the potential to cause abrasion damage and/or mortality to these fish. Trapping of migrants and manual transfer of them downstream over the dam wall may be required during peak migration periods (*i.e.* during autumn floods) if monitoring shows significant injury or mortality due to spillway passage.

A fish screen at the intake would only be required to protect downstream migrating eels if there are power generation turbines installed as part of the outlet. If screening is to be considered, a mesh size of 20 mm and an approach velocity around 0.3 m/s is recommended.

A self-sustaining trout fishery in the reservoir will be reliant on adequate spawning and rearing habitat in the reservoir tributaries, while the size of the fishery will depend on the productivity of the reservoir. The availability of spawning habitat in the tributaries of the upper Lee catchment is largely unknown. It is understood that Fish & Game proposes to resolve this in future studies. As mentioned above, the reservoir is likely to be oligotrophic with relatively low phytoplankton biomass. Therefore, reservoir productivity will depend on the extent to which macrophyte beds establish in the littoral zone around the reservoir margins. The reservoir is likely to have a limited shallow littoral zone, due to the steep sided nature of the valley. This, as well as fluctuations in water level, is likely to limit the development of macrophyte beds to some extent. Those plants that do establish around the shallow margins of the reservoir will periodically be exposed and will probably die off during periods of draw down. However, the incidence of the water level being drawn down to extremely low levels is expected to be relatively rare. Using a relationship between maximum depth limits of rooted macrophytes and observed lake optical properties, macrophytes are expected to establish down to depths of 15 m. Assuming the upper 2.5 m of the littoral zone is discounted (due to its higher frequency of drying), a zone suitable for aquatic macrophyte growth can be expected between approximately 180.4 m and 193.5 m RL. Based on the predicted reservoir surface data available, this area of productive littoral zone is likely to be approximately 32% of the total surface area of the reservoir at maximum capacity.

Excessive macrophyte growth and drifting masses of macrophytes can impair recreational values of reservoirs and clog intakes. A qualitative scoring system developed to assess the risk of excessive aquatic plant growth in reservoirs indicates that there is some risk of macrophytes causing clogging issues, although the risk is toward the low end of the scale. A mitigation plan may be required to address this potential issue. The invasive alga, didymo (*Didymosphenia geminata*) does not generally proliferate in lakes, although masses of senescent cells may be flushed into lakes from growths in tributaries, and then drift into intakes and cause clogging issues. The densely bush-clad nature of the upper Lee catchment means that algal proliferation in the upper catchment is unlikely. However, any management plan aimed at reducing clogging of the outlet with macrophytes is likely to also address any potential clogging issues associated with didymo.

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1. INTRODUCTION

In 2007 Tonkin & Taylor Ltd and its sub-consultants completed a Phase 1 pre-feasibility evaluation of a number of options to provide water storage for long-term irrigation and community supplies in the Waimea Basin, Tasman District. The evaluation was undertaken on behalf of the Waimea Water Augmentation Committee (WWAC). The overall principle of the study was to identify and develop a water augmentation scheme to capture water for storage, and release that water back into the Waimea River system during periods of high water demand and/or low natural water flows to augment those supplies, either directly or via recharging of the groundwater system.

The outcome of that Phase 1 study was to focus feasibility investigations on a water storage dam and reservoir site located in the upper Lee River catchment, a tributary of the Waimea River.

In 2007 WWAC initiated Phase 2 of the study, to take the Lee investigation programme to a feasibility level. This report presents the results of aquatic ecology investigations completed as part of the Phase 2 feasibility study. It is based on a potential dam on the Lee River in Tasman District, at a site approximately 300 metres upstream of the confluence of Anslow Creek and the Lee River. The required storage capacity of the reservoir has been determined to be approximately 13 million m^3, with a normal top water level to Relative Level (RL) 197 m. The proposed dam height is approximately 52 m to the dam crest and the proposed reservoir would extend approximately 4 km upstream from the dam, and cover an area of approximately 65 hectares (based on normal top water level). Figure 1 shows the location of the proposed dam, and the indicative reservoir extent.

In an earlier report for the Phase 1 Waimea Water Augmentation Study (Hay *et al.* 2006), we recommended additional work that would be required to provide information on which to base a consent application for the construction and operation of the scheme. This included:

1. Sampling of water quality and stream invertebrate communities in the vicinity of the dam to ensure that the results from further downstream can be extrapolated to the section of the river under consideration for Phase 2 studies, with the understanding that collection of pre-dam water quality and temperature data would also be useful for any future monitoring efforts;
2. Habitat modelling for the Lee River to determine the effects of the proposed flow regime on habitat availability for key species found in that part of the catchment, and provide a more detailed assessment of an appropriate minimum flow for this reach of the river;
3. An assessment of the flows likely to be required to effectively flush sediment and algae from this reach of the river, in case nuisance periphyton blooms occur below the dam during prolonged periods of low flow.

In this report we provide information on existing water quality and invertebrate communities in the Lee River at the proposed dam site, examine the potential for stratification in the proposed reservoir and implications of different water outlet levels, recommend appropriate minimum and flushing flows for the Lee River downstream of the proposed dam, comment on mitigation of fish passage issues associated with the dam, and address potential issues with reservoir management that may affect fishery values in the reservoir.

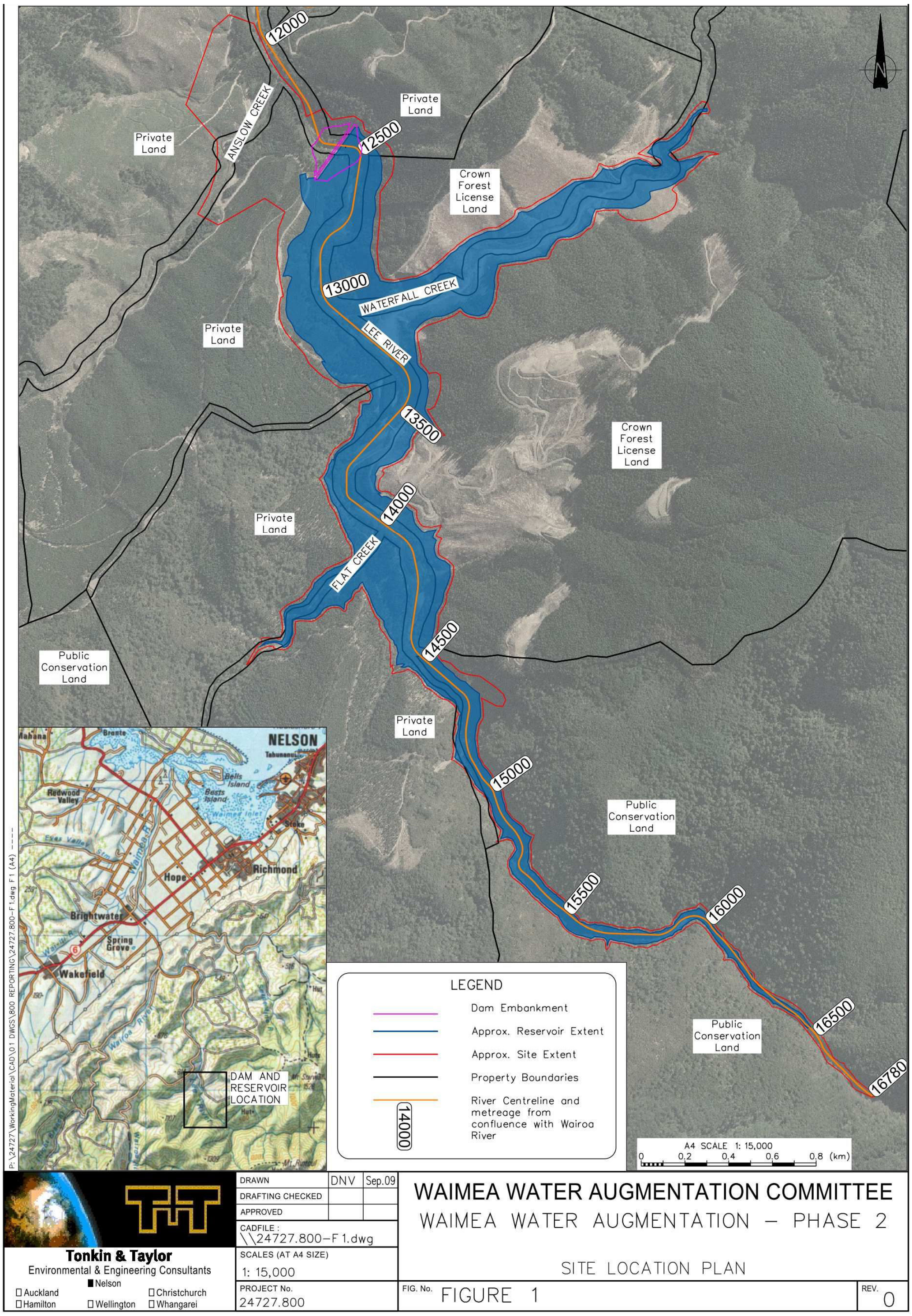

Figure 1. The location of the proposed dam in the Lee Valley and indicative reservoir extent.

2. EXISTING WATER QUALITY AND INVERTEBRATES OF THE LEE RIVER

For the Phase 1 Waimea Water Augmentation Study, Hay & Young (2005b) summarised the information on existing water quality that was available from the Lee River at that stage. However, reconsideration of that data for the Phase 2 Study indicated that it was not clear if the water quality data for the Lee @ Meads Bridge site were equivalent to that upstream at the proposed dam site. There are significant changes in land use surrounding the river between the proposed dam site and Meads Bridge that could potentially cause changes in water quality.

To address this issue for the Phase 2 study, water quality sampling was conducted quarterly on five occasions from January 2008 through to January 2009 at the proposed dam site (described as Lee downstream [ds] Waterfall Creek) in conjunction with the Tasman District Council's (TDC) quarterly State of the Environment (SOE) sampling programme throughout the Tasman District. Parameters measured in the field included dissolved oxygen, temperature, specific conductivity, pH, and black disc water clarity. Samples were collected for analysis of total nitrogen (TN), nitrate-nitrogen (NO_3-N), ammoniacal nitrogen (NH_4-N), total phosphorus (TP), dissolved reactive phosphorus (DRP), faecal indicator bacteria (*E. coli*), dissolved iron (Fe), and dissolved manganese (Mn). Chemical analyses were conducted at the Cawthron Institute laboratories using standard methods. When values were less than the detection limit, a value of half the detection limit was used for comparisons.

Data from the proposed dam site were compared with data available from the Meads Bridge site. Nutrient samples have not been collected by TDC at the Meads Bridge site since 2003, so comparisons were made using the full data set since 2001, rather than between paired samples collected at specific times.

Values for most parameters were very similar between the proposed dam site and the SOE site at Meads Bridge (Figure 2). The only exceptions were concentrations of nitrate-nitrogen and total nitrogen which were elevated at the Meads Bridge site. Concentrations of dissolved iron and manganese have only been measured once at the Meads Bridge site (Fe 0.034 g/m^3, Mn 0.001 g/m^3) limiting the conclusions possible from a comparison between sites. Concentrations of dissolved iron and manganese at the proposed dam site ranged from 0.007-0.022 g/m^3 and 0.0005-0.002 g/m^3, respectively.

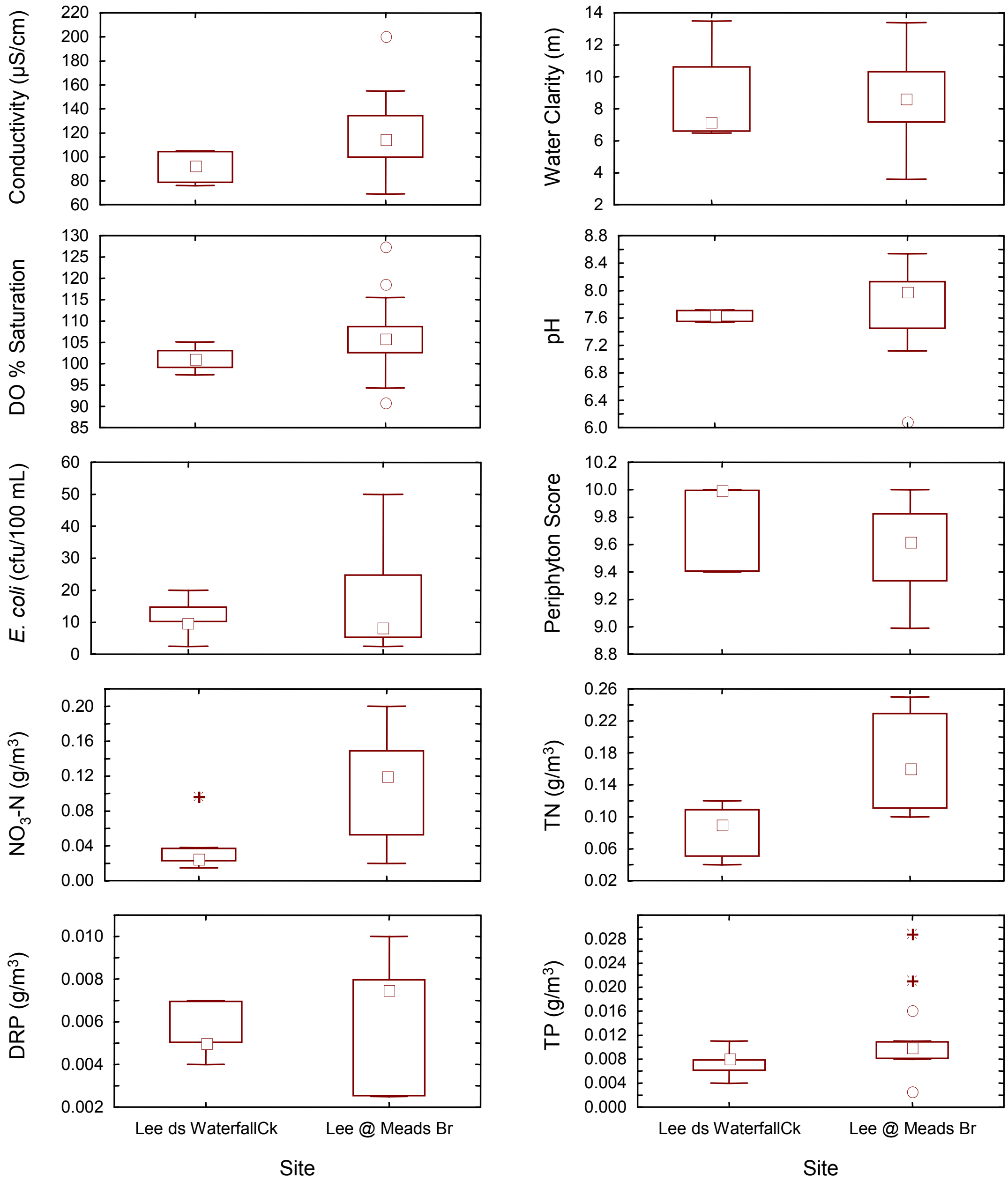

Figure 2. Comparison of existing water quality parameters between the proposed dam site (Lee downstream (ds) Waterfall Creek) and the TDC SOE site (Lee @Meads Bridge). The small squares represent median values. The boxes represent the bounds of the 25th and 75th percentiles of the data. The whiskers represent the non-outlier range. Five data points were available for all parameters at the proposed dam site, while 14-29 data points were available at the SOE site.

Data from both sites in the Lee River were within national guidelines for protection of ecosystem health and compared favourably with other sites in the Tasman District (Table 1; Young *et al.* 2005).

Table 1. Existing median water quality values at the sampling sites in the Lee River compared with water quality guidelines

Parameter	Guideline	Lee ds Waterfall Ck	Lee @ Meads Br
DO Saturation (%)	>80	101	106
E. coli (cfu/100 mL)	<260	10	8.5
NH_4-N (g/m^3)	<0.01	0.006	0.0065
NO_3-N (g/m^3)	<0.167	0.025	0.12
TN (g/m^3)	<0.295	0.09	0.16
DRP (g/m^3)	<0.009	0.005	0.0075
TP (g/m^3)	<0.026	0.008	0.01
Turbidity (NTU)	<4.1	0.3	0.4
Water Clarity (m)	>1.6	7.2	8.7
Iron (g/m^3)		0.01	0.034
Manganese (g/m^3)	<1.2	0.0011	0.001

No dramatic differences in the invertebrate community in the Lee River between the proposed dam site and the Meads Bridge site were evident (Table 2). Mayflies, stoneflies and caddis flies, which are typical of healthy river systems, dominated the invertebrate community at both sites (Table 2). Indices used to indicate the health of river systems were consistent between sites with SQMCI and MCI scores above 7 and 125, respectively, reflecting very good health, except during the drought of 2001 when invertebrate communities were more indicative of satisfactory/good ecosystem health. No rare species were observed at either site, although the taxonomic resolution of invertebrate identification would have to be conducted at a higher level to definitively confirm this.

Table 2. Relative abundance of existing invertebrate communities in the Lee River at Meads Bridge and the proposed dam site (Lee ds Waterfall Ck). Abundances classes are: Rare (1-4, R); Common (5-19, C); Abundant (20-99, A); Very Abundant (100-499, VA); Very Very Abundant (>500, VVA)

Site	Lee @ Meads Br Drought	Lee @ Meads Br	Lee @ Meads Br	Lee @ Meads Br	Lee ds Waterfall Ck
Date	**11-Apr-01**	**04-Oct-01**	**14-Oct-02**	**10-Nov-03**	**14-May-08**
Mayflies					
Austroclima jollyae					R
Coloburiscus humeralis		R		R	A
Deleatidium spp.	A	VA	A	VVA	VA
Nesameletus spp.			C	R	A
Stoneflies					
Stenoperla prasina		R			C
Stenoperla sp.		R			
Zelandoperla decorata				R	A

Site Date	Lee @ Meads Br Drought 11-Apr-01	Lee @ Meads Br 04-Oct-01	Lee @ Meads Br 14-Oct-02	Lee @ Meads Br 10-Nov-03	Lee ds Waterfall Ck 14-May-08
Caddis flies					
Aoteasyche spp.	R	C	C	A	VA
Beraeoptera roria		R	C	A	
Confluens olingoides		R			R
Costachorema psaropterum				R	
Costachorema sp.		R		R	R
Costachorema xanthopterum		R	R		
Helicosyche sp.		R	C	C	R
Hudsonema amabilis	R				
Hydrobiosis clavigera		R			
Hydrobiosis copis					R
Hydrobiosis parumbripennis		R			
Hydrobiosis sp.					R
Olinga feredayi	A	C	A	C	VA
Psilochorema leptoharpax	R	R			
Psilochorema macroharpax					R
Pycnocentria evecta	R				
Pycnocentrodes sp.	VA	A	C	VA	R
Dobsonflies					
Archichauliodes diversus	C	A	R	C	C
Beetles					
Elmidae	VA	C	C	C	A
Hydraenidae		R		R	A
Ptilodactylidae					R
Flies (Diptera)					
Aphrophila neozelandica		A	C	C	A
Austrosimulium spp.				R	C
Eriopterini	C	C	R		R
Maoridiamesa spp.	R	C	R		
Orthocladiinae	R	R	R	R	R
Parochlus sp.		R			
Polypedilum sp.					R
Tabanidae		R			
Tanytarsus sp.	R			R	
Snails					
Potamopyrgus antipodarum	A				C
Acarina			R	R	
Worms (Annelida)	R				
Number of taxa	15	24	15	19	25
EPT Taxa	7	15	8	11	15
SQMCI	5.88	7.00	7.51	7.39	7.16
MCI	109	132	129	129	136

3. POTENTIAL RESERVOIR STRATIFICATION AND WATER QUALITY

Most lakes deeper than 20-30 m will tend to stratify during summer with a warmer surface layer 'floating' above a higher density, cooler layer. A substantial amount of energy is required to break up this thermal stratification and therefore there is often little mixing between the surface and bottom waters during periods of stratification. As winter approaches, temperatures of the surface water decline reducing the difference in density between surface and bottom waters. Complete mixing of the water column is then more likely, in association with wave action and river inflows.

During periods of stratification, oxygen demand in the bottom of a lake can lead to reductions in dissolved oxygen concentration in the bottom waters because they are isolated from reaeration through the lake surface. In severe cases oxygen can be removed completely creating anoxic conditions that can not be tolerated by most higher organisms. Anoxia can also lead to further water quality problems, such as the release of dissolved phosphorus and dissolution of iron and manganese from sediments. These problems are likely to be most apparent over the first five years after a reservoir is filled while inundated vegetation and soils are decomposed.

The proposed reservoir has a predicted capacity of 13 million m^3, normal top water level at approximately 197 m RL, maximum surface area of 0.63 km^2, mean inflow of 3.8 m^3/s, and water residence times of 40 days. The base of the reservoir will be at 150 m RL with water depth up to 46 m. It is assumed that most vegetation will be removed from the reservoir footprint prior to filling.

To assess the likely potential for stratification in the proposed reservoir we used a computer model (DYRESM – Dynamic Reservoir Simulation Model) developed by the University of Western Australia. This model has been widely adopted for similar purposes throughout the world. More information and supporting documents about the model are available at http://www.cwr.uwa.edu.au/services/models.php .

To run the model, information on climate (incoming solar radiation, cloud cover, air temperature, humidity, wind speed, rainfall), reservoir location and morphometry (latitude, altitude, valley slope, dam height, outlet heights, bathymetry), inflow volumes and temperature, and outflows are required. Predicted inflows, outflows, and reservoir morphometry were supplied by Tonkin & Taylor Ltd (Lee Dam Operating Regime 26 March 2009 No Hydro.xls: David Leong). Climate data were obtained from weather stations at Nelson Airport via the National Institute of Water and Atmospheric Research (NIWA)'s Cliflo database over the period from 1 January 1999 to 31 December 2007. Daily average inflow temperatures over the same period were derived from a correlation between daily average air temperatures and observed daily average water temperatures ($R^2 = 0.81$) at the proposed dam site from 17 January 2008 to 26 March 2009. The model then simulated reservoir level, spill flows and water temperatures throughout the water column at the deepest point in the reservoir. The effect of different water outlet heights on stratification characteristics within the reservoir was determined with outlet levels at 155 m RL, 160 m RL and 165 m RL. Model runs with higher outlet levels were not conducted because water levels in the reservoir were predicted to drop to 166 m RL at times during the simulation period.

The model outputs show a general pattern of cool temperatures throughout the water column during winter indicating a well mixed water column (Figure 3). Stratification occurs in late spring, summer and autumn with the thermocline approximately 10 m below the reservoir surface during wet years (1999/2000, 2001/2002, 2004/2005) and much deeper (30+ m below the surface) during dry years (2000/2001; 2005/2006). Water temperatures in summer are predicted to be in the mid-20s (°C) near the surface of the reservoir, but below 10°C near the bottom (Figure 3).

The level selected for the outlet is predicted to have a large influence on where the stratification layer will form in the reservoir (Figures 3, 4 and 5). A low level release outlet near the base of the dam (155 m RL, Figure 5) would result in a fully mixed water column during summer in dry years (*e.g.* 2000/2001). However, during normal or wet years an outlet at this level would potentially be releasing any poor quality bottom water. An outlet approximately 10 m below the reservoir surface (185 m RL) would ensure that it was possible to release good quality surface water from the reservoir during most years, although in dry periods the reservoir may drop below 185 m RL making this higher level outlet redundant. Hydrological modelling for the period from 1957 to 2007 indicates that the reservoir would drop below 185 m RL around 1% of the time. Given these results we recommend that two outlet levels are incorporated into the scheme design. One outlet should be near the bottom of the reservoir to be used during dry years and during floods to flush any poor quality bottom water from the reservoir. A second outlet should be provided at approximately 185 m RL to be used under most conditions unless reservoir levels were too low. Given that water temperatures near the surface of the reservoir will be relatively high at times, it may also be possible to manage outlet use to limit thermal effects on the river downstream. This strategy is currently used at the Maitai Reservoir to limit changes in water temperature as a result of discharges from the dam.

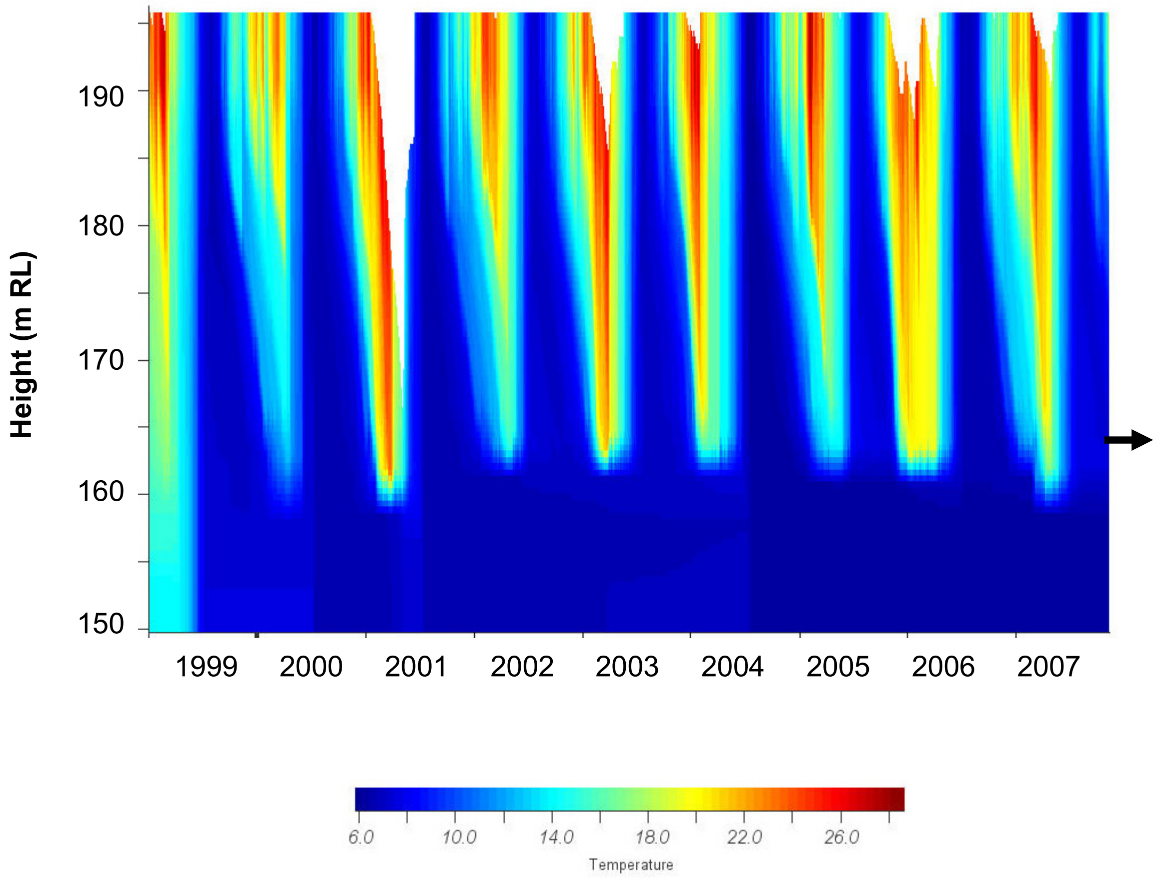

Figure 3. Predicted temperature in the reservoir 1999-2007 with low level release at 165 m RL (shown with arrow).

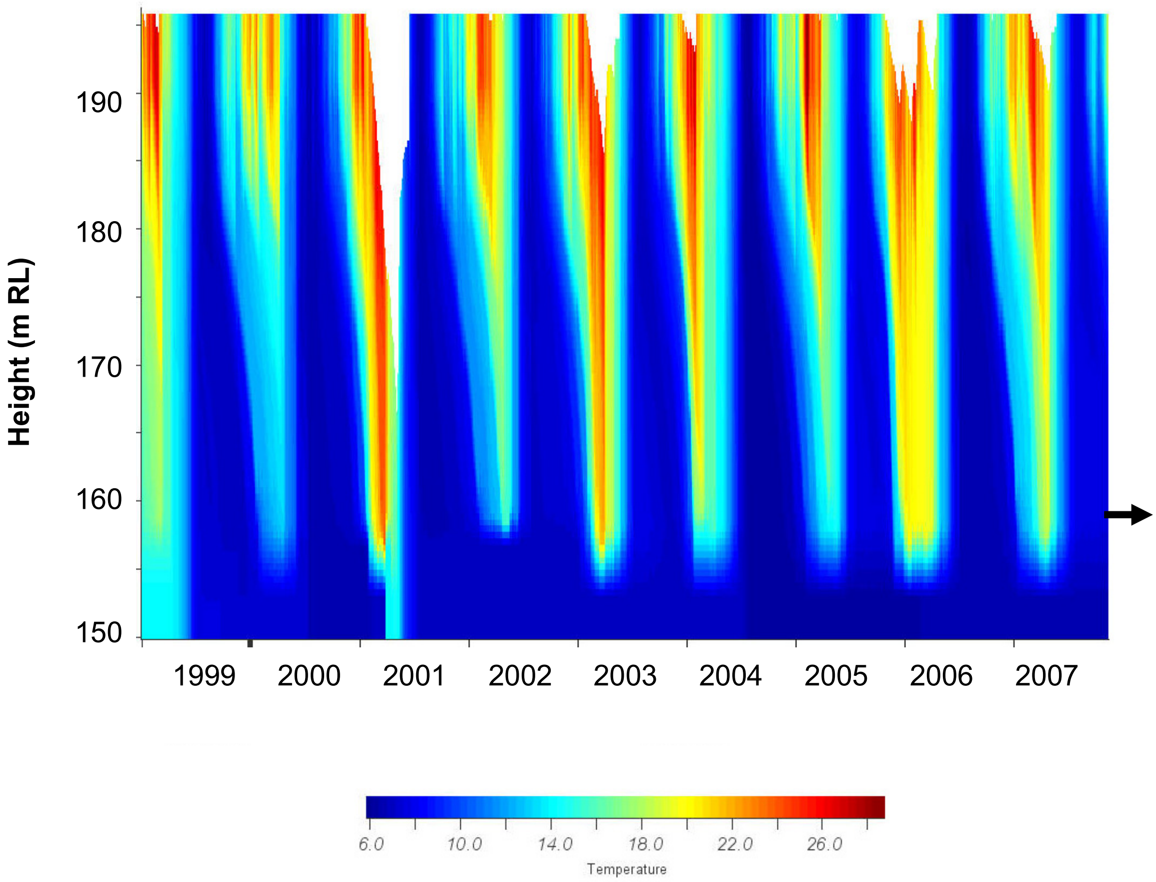

Figure 4. Predicted temperature in the reservoir 1999-2007 with low level release at 160 m RL (shown with arrow).

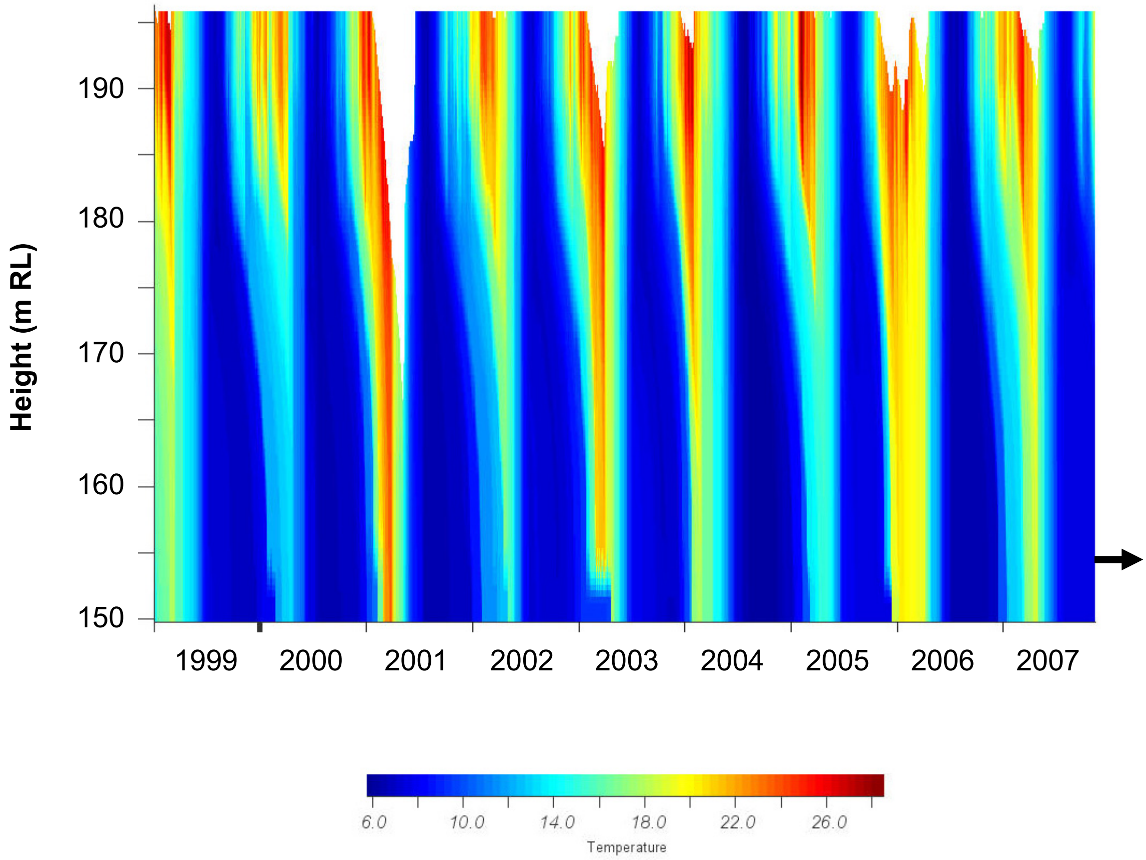

Figure 5. Predicted temperature in the reservoir 1999-2007 with low level release at 155 m RL (shown with arrow).

4. PREDICTED TROPHIC STATUS OF THE RESERVOIR

Predictions of the trophic status of lakes can be made based on empirical relationships between nutrient concentrations and observed concentrations of algal biomass and water clarity in lakes. Burns *et al.* (2000) developed a 7-tier classification scheme for lakes in New Zealand with clear low-nutrient lakes referred to as ultra-microtrophic while turbid high nutrient lakes are described as hypertrophic (Table 3). Mean concentrations of total nitrogen and total phosphorus at the proposed dam site are 0.082 gN/m^3 and 0.0074 gP/m^3, respectively. Assuming that current nutrient loading in the Lee River will remain the same after the construction of the reservoir, these nutrient concentrations would indicate that the reservoir would be oligotrophic with chlorophyll *a* concentrations expected to be below 2 mg/m^3 and water clarity (Secchi dish) between 7-15 m (Table 3).

As mentioned in Section 2 above, there is a relatively limited dataset for nutrient concentrations at the proposed dam site. Over a longer time period, the mean concentrations of total nitrogen and total phosphorus at the Lee River @ Meads Bridge site are 0.17 gN/m^3

and 0.01 gP/m^3, respectively. If these concentrations are a more accurate reflection of the nutrient load, the reservoir would be mesotrophic with chlorophyll *a* concentrations between 2-5 mg/m^3 and water clarity between 2.8-7 m (Table 3).

Table 3. Values of key variables that define the boundaries between different trophic levels in lakes, from Burns *et al.* (2000).

Lake type	Trophic level index (TLI)	Chl-*a* (mg m^{-3})	Secchi disk (m)	TP (mg P m^{-3})	TN (mg N m^{-3})
Ultra-microtrophic	0 – 1	0.13 – 0.33	33 – 25	0.84 – 1.8	16 – 34
Microtrophic	1 – 2	0.33 – 0.82	25 - 15	1.8 – 4.1	34 – 73
Oligotrophic	2 – 3	0.82 – 2.0	15 - 7	4.1 – 9.0	73 – 157
Mesotrophic	3 – 4	2.0 – 5.0	7 – 2.8	9.0 – 20.0	157 – 337
Eutrophic	4 – 5	5.0 – 12.0	2.8 – 1.1	20 – 43	337 – 725
Supertrophic	5 – 6	12.0 – 31	1.1 – 0.4	43 – 96	725 – 1558
Hypertrophic	7 – 8	>31	<0.4	>96	>1558

The above predictions of lake trophic status are based on predicted loads of nutrients from inflows. Nutrients contained within remaining vegetation and soils inundated by the reservoir will potentially be released into the water column over the first few years after the reservoir is filled. The effects of this extra nutrient source will potentially boost phytoplankton productivity, but this effect is expected to decline over time.

The residence time of water in the proposed reservoir will also influence the likelihood of phytoplankton blooms. For example, Pridmore & McBride (1984) noted that residence times of less than 14 days may limit phytoplankton growth even when nutrients are not limiting. For the Lee Dam, modelling predicts an average residence time in the proposed reservoir of 40 days. Accordingly, nutrient limitation will be an important factor controlling phytoplankton blooms and the trophic status of the lake.

Our predictions that the proposed reservoir in the Lee Valley will be oligotrophic match observations from the neighbouring Maitai Reservoir (Stark 2000). Concentrations of total phosphorus in the Maitai Reservoir are low (<0.03 gP/m^3) and chlorophyll *a* concentrations are normally less than 2 mg/m^3 (Stark 2000). Given the similarity in climate and land-use in the catchment above the Maitai and proposed Lee Reservoir, support for our predictions is reassuring.

5. INSTREAM HABITAT

During the Phase 1 pre-feasibility study a range of potential minimum flows were suggested for the Waimea River at the Appleby Bridge (Hay & Young 2005a). The WWAC selected a minimum flow of 1100 l/s for the Waimea at Appleby after all abstractions have been taken into account. This corresponds to the natural 1-day MALF and should maintain habitat availability at or above the level that would have been expected without any abstractions in most years. In the Phase 1 study Hay & Young (2005a) also recommended maintenance of the existing mean annual low flow (MALF; 470 l/s[1], estimated 7-day MALF) as the environmental benchmark minimum flow immediately below the potential dam site, and this was adopted in the pre-feasibility modelling.

During the current Phase 2 feasibility study, habitat modelling was conducted to provide a more robust indication of the instream habitat requirements at the proposed dam site, and support a more scientifically defensible minimum flow decision to maintain instream habitat in the Lee River below the dam site.

The hydraulic model constructed for this purpose was also used to assess the magnitude of flushing flows required to remove excessive periphyton build-ups, should these occur (as discussed in Section 6).

5.1. Methods

Flow requirements for instream habitat in the Lee River were assessed by Instream Flow Incremental Methodology (IFIM) habitat modelling with the computer programme RHYHABSIM version 5.0 (developed by I. Jowett, NIWA).

5.1.1. Habitat modelling within the IFIM

The IFIM is a decision-support system (or framework), which provides a process for solving water allocation problems where there are concerns for maintaining instream habitat (Bovee *et al.* 1998). Within this process, computer modelling of instream habitat availability for selected species (or suitable depths and velocities for given aquatic activities), over a range of flows, provides a basis for decision making regarding allocation of water resources.

Habitat modelling within the IFIM entails measuring water depths and velocities, as well as substrate composition, across several representative stream cross-sections at a given flow (referred to as the survey flow). Points on the banks, above water level, along the cross-sections are also surveyed to allow model predictions to be made at flows higher than the survey flow. The stage (water level) at zero flow is also estimated at each cross-section to facilitate fitting of rating curves and for making model predictions at low flows. Other data for

[1] Note this estimate of MALF has more recently been increased to 511 l/s on the basis of additional flow data, and this higher estimate has been used as the minimum flow for hydrological modelling of the post-scheme scenario below the dam.

fitting rating curves are obtained from additional measurements of water level at each cross-section, relative to flow, on subsequent visits. These data allow calibration of a hydraulic model for predicting how depths, velocities and the area of different substrate types covered by the stream will vary with discharge in the surveyed reach.

Modelled depths, velocities and substrate types can then be compared with habitat suitability criteria (HSC) describing the suitability of different depths, velocities and substrate sizes as habitat for given species of interest. These criteria take the form of habitat suitability curves, which have been developed by observing the depths and velocities used by various species, both in New Zealand and overseas. Comparison of the HSC with the modelled physical characteristics of the study stream provides a prediction of the availability of habitat in the stream. Habitat modelling is undertaken over a range of flows to predict how habitat availability will change with flow.

The modelled depths and velocities can also be used to assess flow requirements to maintain fish passage by looking at changes in water depth and water velocity on shallow riffle sections.

5.1.2. Weighted Usable Area - the currency of flow decision making

Modelled habitat availability is expressed as an index called Weighted Usable Area (WUA), which is calculated as the sum of the area weighted products of the combined habitat suitability scores (*i.e.* depth x velocity x substrate suitabilities) for the measurement points on the cross-sections. Traditionally WUA has often been expressed as an area per linear metre of river reach (m^2/m). However, WUA is actually a dimensionless index providing an indication of the relative quantity and quality of available habitat predicted at a given flow. Predicted changes in habitat quantity and habitat quality are integrated in WUA.

Traditionally there has also been an alternative expression of WUA as a percentage. This was intended to provide an indication of the quality of predicted habitat (I. Jowett, NIWA, pers. comm.). However, it has frequently been interpreted as another quantitative metric, indicating the percentage of the reach that will provide suitable habitat at a given flow. This metric has been changed in recent versions of RHYHABSIM (Version 3.31 and above) to a Habitat Suitability Index (HSI, ranging between 0 and 1) in an attempt to reduce confusion around interpretation. This metric is the average combined habitat suitability score taken over the modelled reach and is intended to provide an indication of the relative quality of the predicted available habitat (I. Jowett, NIWA, pers. comm.).

It is important to realise that these metrics provide only a relative measure of how predicted habitat changes with flow. Therefore, when interpreting the WUA x flow or HSI x flow curves that are the output of modelling, it is the shape of the curves (*e.g.* the flows at which the optimum WUA and major changes in slope occur) that are of interest, rather than the magnitude (or height) of the WUA x flow curves (although the magnitude of HSI is more directly comparable between rivers). These outputs provide an indication of how habitat availability is predicted to change with flow. WUA serves as a currency which stakeholders can use for interpreting effects of flow change on instream habitat and for negotiating flow decisions.

All of the predicted habitat x flow figures referred to in this report show the WUA metric. However, graphs of Habitat Suitability Index (HSI; the equivalent of WUA% in earlier versions of RHYHABSIM) versus flow are attached for completeness (Appendix 2). The HSI x flow curves are generally similar in shape to the WUA x flow curves, although the former often peak at lower flows. Flow decisions based on the WUA x flow curves are therefore likely to be more conservative.

5.1.3. Reach selection for IFIM habitat modelling

There are two approaches that can be followed when selecting locations for the cross-sections, which form the basis of the field survey component of habitat modelling; habitat mapping or the representative reach (Jowett 2004). In the habitat mapping approach the proportion of each habitat type (*e.g.* run, riffle, pool) comprising a relatively long reach of the stream is mapped and each cross-section is given a percentage weighting based on the proportion of the habitat in the reach that it represents. The predictions of subsequent modelling then relate to the reach that was mapped.

In the representative reach approach a relatively short (typically 50-150 m over at least one riffle – run – pool sequence) reach of river is selected that is thought to be representative of a longer section of river (Jowett 2004). The cross-sections are closely spaced (at a scale of metres) at longitudinal points of habitat change along the reach, with note being taken of the distance between cross-sections, and water levels on all cross-sections being surveyed to a common datum. The subsequent modelling predictions are then assumed to be applicable to the section of river that the chosen reach represents.

Whichever of these approaches is employed, the underlying assumption is that the cross-sections measured provide a reasonable representation of the variability in habitat throughout the reach of interest.

The number of cross-sections required depends on the morphological variability within the river. Studies have shown that relatively few cross-sections can reproduce the shape of the WUA – flow relationship obtained from a survey with a large number of cross-sections:

- Milhous (1990) visually compared results from sub-samples of four transects (one per sampling unit) selected from a set of 24 transects and, with some minor reservations, concluded that “the shape of the relationships are similar…” and the “number of cross sections can be relatively small…”.
- In a study of 86 study sites on 58 Wisconsin streams, Simonson *et al.* (1994) found that 20 transects gave means accurate to within 5% of the true mean 95% of the time. With 13 transects, 85% of the means were within 5% of the true means.

- Bovee (1997) concluded that pocket water, a complex habitat type containing a wide variety of depths and velocities, can be accurately described with three to five transects.
- Payne *et al.* (2004) sub-sampled several very large data sets to determine how many cross-sections were required to produce a robust WUA function, and found that 18-20 cross-sections gave results nearly identical to results for 40-80 cross-sections per reach and only a few cross-sections were required to reproduce the general shape of the relationship.

The total number of cross-sections needed to generate a robust result should be proportional to the complexity of the habitat hydraulics: 6-10 for simple reaches and 18-20 for diverse reaches.

5.1.4. Field data collection

The habitat mapping approach was applied in the Lee River. Two survey reaches were initially selected on the basis of channel slope and tributary inflows (Figure 6):

1. The reach immediately below the initial proposed dam site extending into the top end of a steep gorge around Lucy Creek (Figure 6). This reach was intended to represent habitat conditions from the initial proposed dam site into the steep gorge section below Lucy Creek (Figure 6).
2. The lower gradient section above Meads Bridge. This reach was intended to represent habitat conditions from the bottom of the steep gorge section below Lucy Creek to the confluence with the Roding River (Figure 6).

However, the dam site was subsequently moved due to geotechnical considerations and the upper study reach was extended to account for this (Figure 6).

The initial habitat mapping and cross-section selection were carried out by Cawthron, Fish & Game, Department of Conservation and Tasman District Council staff on 28 February 2008. Approximately a 1.9 km long reach was initially mapped in the Lucy Creek reach and approximately a 1.65 km reach in the Meads Bridge reach. An additional 1.3 km was habitat mapped in the Anslow Creek reach, above the Lucy Creek reach, by Cawthron and Fish & Game staff on 26 March 2009, following the decision to move the proposed dam site upstream. The channel in the Anslow Creek reach was less confined than that in the Lucy Creek reach and five additional cross-sections were located in the Anslow Creek reach so that this morphological difference would be represented in the final habitat modelling dataset. Cross-sections and habitat mapping data from the Lucy Creek and Anslow Creek reaches were combined into a single dataset for analysis, referred to as the Upper reach (Figure 6).

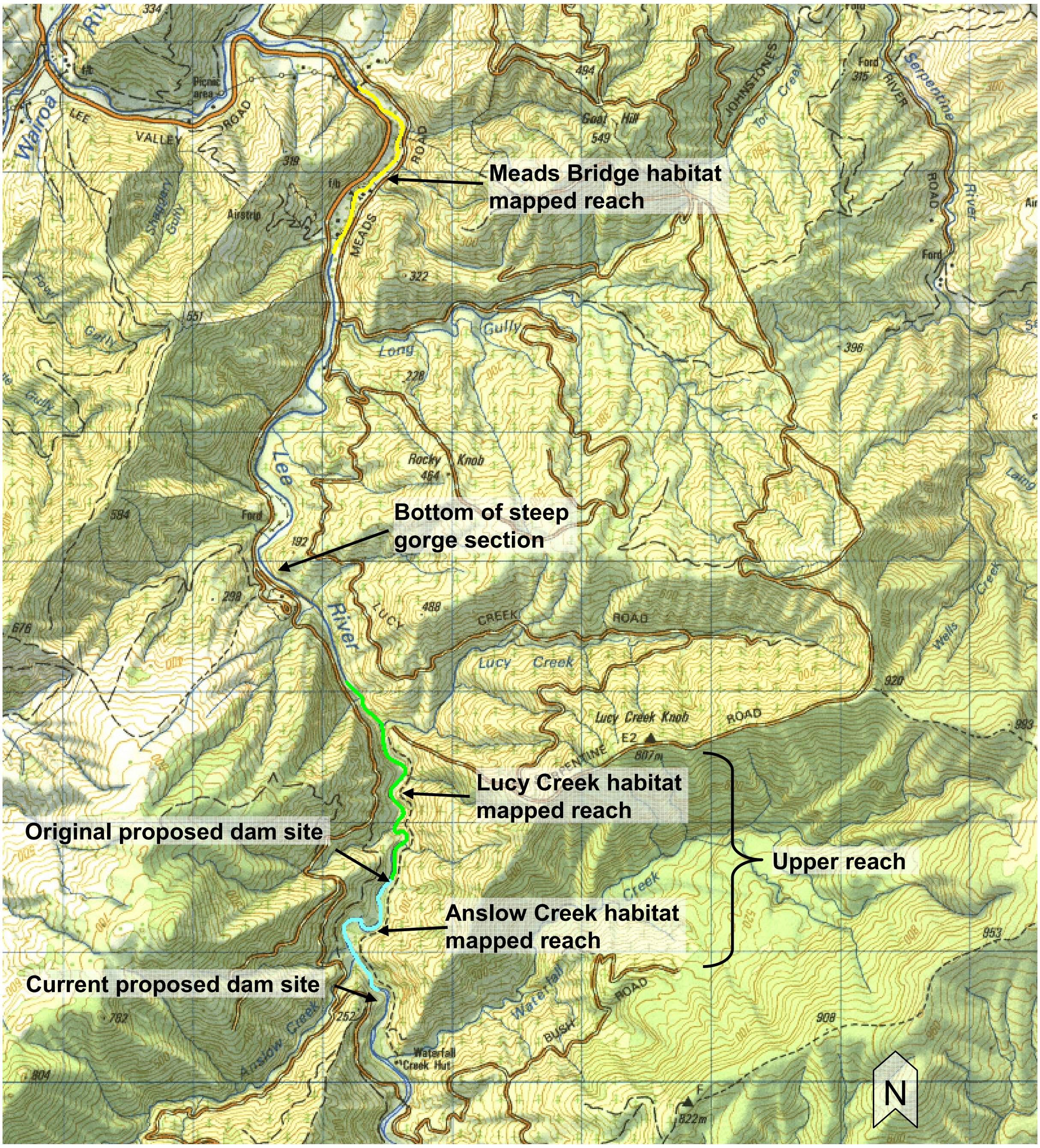

Figure 6. The Lee River, showing the reaches on which the IFIM habitat analyses were based, and the original and current proposed dam locations.

Five meso-habitat types were identified within the Lucy Creek reach and the Anslow Creek reach (Table 4), while four meso-habitat types were identified the Meads Bridge reach (Table 4). Within each habitat type, cross-sections were positioned in an attempt to encompass the full range of variability represented in each of these habitat types.

All subsequent field work for the Meads Bridge and Lucy Creek reaches was undertaken over the month following initial habitat mapping, with the main cross-section habitat surveys undertaken on 27 March 2008. The flow at the time of the habitat surveys was 0.681 m^3/s in the Lucy Creek reach and 0.759 m^3/s in the Meads Bridge reach. The calibration water level readings for the Anslow Creek reach were taken between 26 March and 27 April 2009, with the main cross-section habitat survey undertaken on 16 April 2009 at a flow of 0.504 m^3/s.

Stage – discharge relationships for each cross-section were developed based on three measurements in addition to the gauging and cross-section water level measurements at the survey flow. The calibration water level measurements were collected at a range of flows between 0.681 m^3/s and 2.210 m^3/s flow range in the Lucy Creek reach, between 0.504 m^3/s and 4.045 m^3/s in the Anslow Creek reach, and between 0.759 m^3/s and 2.243 m^3/s in the Meads Bridge reach.

Within RHYHABSIM, the default survey flow for calibration is the average of the flows gauged at all the cross-sections during the survey of depths and velocities. However, this can be unduly affected by outlier estimates from cross-sections on which accurate gauging is difficult (*e.g.* those located in turbulent riffles or deep pools). For this reason the survey flow was specified based on the best available estimate of flow in the reach at the time of the survey (Martin Doyle, TDC hydrologist, pers. comm.).

For habitat modelling analysis the Upper reach and Meads Bridge reach were combined and analysed together. This resulted in a dataset consisting of 27 cross-sections representing the range of habitat types in the river from immediately below the new proposed dam site to the Roding River confluence.

For flushing flow analysis the Upper reach and Meads Bridge reach were analysed separately. The flushing flow analysis is strongly dependent on the river gradient, and these two sections of the river have quite different gradients.

Table 4. Summary of habitat mapping and cross-section allocation in Lee River IFIM reaches.

Reach	Habitat type	Percentage of total reach length (%)	Percentage of reach excluding cascade (%)	Number of cross-sections	Weighting per cross-section (%)
Upper	Pool	37	37	5	7.43
(Anslow Creek &	Riffle	16	17	3	5.54
Lucy Creek reaches	Run	36	37	5	7.36
combined)	Pocket-water	9	9	2	4.73
	Cascade	2			
	Total			15	
Meads Bridge	Pool	31	33	4	8.25
	Riffle	18	19	3	6.33
	Run	46	48	5	9.60
	Cascade	5			
	Total			12	

5.1.5. Data checking

The data sets were imported into RHYHABSIM and checked to ensure that they met expectations of data quality. Aside from the standard checks performed within the programme's built in data checking function:

- Cross-sections were plotted and visually checked for any obvious anomalies (*i.e.* unrealistic depth and velocity spikes).
- Rating curves were checked to see that they exhibited a good fit to the expected power curve relationship, and that the different types of rating curves calculated in RHYHABSIM did not substantially differ from one another.
- The Velocity Distribution Factors (VDFs) were edited so that points falling above the water surface at the survey flow were given reasonable VDF values (*i.e.* vary around a value of one, and generally decrease with distance toward the banks (Jowett 2004); the default is that they are given the same value as the closest point which was below water level at the survey flow). This consideration is important when modelling flows above the survey flow.

In general the data sets for both reaches met most data quality expectations, aside from a few data entry errors, which were corrected when found. There were two other minor issues with the data sets:

1. The recorded percentage cover of different substrate types at a few points did not sum to 100%. In these instances reasonable proportions of each substrate type were interpolated based on adjacent data points such that they summed to 100%.
2. The field records for depths of two points in the Anslow Creek reach appeared to be an order of magnitude too large. These depths were probably recorded in the field in cm rather than m, and the data for these two points were adjusted accordingly to remove the unrealistic spikes in depth.

Discharge was assumed to be constant between cross-sections within each reach for all of the modelling predictions (*i.e.* there was assumed to be no significant inflow *e.g.* from tributaries or groundwater, and no significant losses *e.g.* to groundwater or abstraction, over the length of each modelled reach). Tributary inflows between the Upper reach and the Meads Bridge reach were accounted for in the modelling using flow correlations provided by TDC[2].

[2] The flow correlations used were:
Lee between Anslow Creek and Lucy Creek (l/s) = Lee above Waterfall Creek (l/s) x 1.168 – 20,
and,
Lee at Meads Bridge (l/s) = 1.2341 x Lee above Waterfall Creek (l/s) + 65.6
(the latter correlation was based on only four gauging pairs).

5.1.6. *Habitat modelling*

Habitat Suitability Criteria

The selection of species to include in habitat modelling was based on the species recorded from the Lee Catchment in the New Zealand Freshwater Fish Database (NZFFD 2009). On 5 May 2009 the NZFFD contained records of seven species of fish from the catchment (including one exotic species), and one species of crustacean (Table 5). A review of biological data from the Waimea catchment (reported in Hay & Young 2005b) included a single record of an additional species in the Lee Catchment, a shortfin eel from Anslow Creek. This record was based on an observation made by DOC in January 2005, which does not appear to have been included in the NZFFD.

Table 5. Fish species recorded from the Lee catchment in the New Zealand Freshwater Fisheries Database. Database accessed on 5 May 2009.

	Common name	Scientific name	Number of records	Habitat modelled
Native fish	Longfin eel	*Anguilla dieffenbachii*	8	Habitat for two size classes modelled
	Shortfin eel[†]	*Anguilla australis*	1	Habitat for one size class modelled
	Torrentfish	*Cheimarrichthys fosteri*	1	Habitat modelled
	Koaro	*Galaxias brevipinnis*	2	Habitat modelled
	Bluegill bully	*Gobiomorphus hubbsi*	1	Habitat modelled
	Redfin bully	*Gobiomorphus huttoni*	4	Habitat modelled
	Upland bully	*Gobiomorphus breviceps*	2	Habitat modelled
Exotic fish	Brown trout	*Salmo trutta*	4	Adult and yearling and spawning habitat modelled
Crustacea	Koura	*Paranephrops planifrons*	3	Excluded - No HSC available

[†] Not recorded in the NZFFD. Recorded by DOC from Anslow Creek in January 2005.

As illustrated by the number of records in Table 5, fish data for the Lee River are sparse. However, the species recorded are likely to be a reasonable representation of those present.

Predicted changes in physical habitat with flow were modelled for all of those species from Table 5 for which habitat suitability criteria (HSC) were available. Appendix 1 provides graphical representations of the suitability criteria applied, and their sources.

IFIM habitat modelling predictions are most sensitive to the habitat suitability criteria applied (Jowett 2004). Therefore, the HSC chosen for a study must be appropriate for the species which are known to (or are likely to) occur in the study river. When several different sets of HSC are available for a given species (as is the case with brown trout) the suitability criteria should be selected to best represent the habitat needed to maintain a population of the species of interest. Consideration must also be given to the transferability of HSC developed on other rivers to the study river. It seems reasonable to expect that HSC developed on rivers with similar physical characteristics to the study river should be more readily applicable, than HSC developed on physically different rivers.

Hayes & Jowett's (1994) suitability criteria have been used most widely in New Zealand for modelling adult drift-feeding brown trout habitat since their development. These HSC were developed based on observations of habitat preferences of large (45–65 cm) actively feeding brown trout on moderate-sized rivers (upper Mataura, Travers, upper Mohaka) over the flow range 2.8–4.6 m^3/s. This flow range is higher than generally experienced during periods of low flow in the Lee River in the sections of interest (Natural 7-day MALF 0.511 m^3/s at the proposed dam site, based on synthesised data for the period of record July 1958- June 2007 for this site (flow data provided by Tonkin &Taylor Ltd), which would equate to approximately 0.590 m^3/s in the Meads Bridge reach). This may cause these criteria to overestimate the flow requirements of trout in the modelled reaches of the Lee. However, the channel gradient in the river sections that the surveyed reaches in the Lee were intended to represent falls within the range of gradients in Hayes & Jowett's (1994) study rivers (approximately 0.0145 m/m in the Upper reach, and 0.0056 m/m in the Meads Bridge reach *c.f.* 0.0016-0.0211 m/m in Hayes & Jowett's study rivers). Consequently, average velocities may not be too dissimilar despite the lower flow range.

The HSC for juvenile brown trout (15-25 cm) developed by Raleigh *et al.* (1986) have been used extensively in New Zealand IFIM habitat modelling applications in the past, although they may underestimate flow requirements due to the inclusion of resting fish observations in the development of the criteria, which tends to give them a bias toward slower water habitat (a common problem in older habitat suitability criteria; Hayes 2004). For this reason the criteria developed by Roussel *et al.* (1999) and Hayes & Jowett (1994) may provide a more conservative estimate of habitat availability, since they were based solely on observations of actively feeding fish. As with Hayes & Jowett's (1994) criteria, those developed by Raleigh *et al.* (1986) were based on observations over a higher flow range (1.1-7.8 m^3/s), but a similar river gradient to the pertinent sections of the Lee River.

Roussel *et al.* (1999) have also developed HSC for juvenile to small adult brown trout, taking care to only include actively feeding fish, and taking account of the constraints of habitat availability. These HSC, developed in France, have not been widely applied in New Zealand before. However, they may be the best available HSC for application to the Lee River because they were developed on a stream of comparable size (*i.e.* 110 l/s during the observations on which the HSC were based, *c.f.* mean annual low flow of 470 l/s in the Lee at the proposed dam site), although the stream gradient was slightly lower (approximately 0.0145 m/m in the Upper reach, and 0.0056 m/m in the Meads Bridge reach *c.f.* 0.001 m/m in the study stream used by Roussel *et al.* 1999). However, the suitability criteria for substrate from Roussel *et al.* (1999) do not comply with expectations, based on both experience and the weight of evidence in the literature. While it is generally accepted that juvenile brown trout are associated with coarse substrate (cobbles and boulders), the substrate criteria in this set of HSC showed them to prefer fine sandy substrate. This anomaly may have been caused by the larger substrate elements being embedded in sandy substrate, in the stream where these criteria were developed. For this reason these HSC were applied with the effect of substrate removed in these analyses.

Although it is not known whether much spawning occurs in the mainstem Lee River in these reaches, it is likely that the majority of spawning activity occurs in tributaries or further upstream above the modelled reach. However, trout spawning habitat requirements were modelled for completeness. Shirvell & Dungey (1983) developed HSC for brown trout spawning in New Zealand rivers and these criteria have been widely used in New Zealand IFIM habitat modelling applications. However, Shirvell & Dungey's velocity suitability criteria are based on near bed velocities rather than mean column velocities (*i.e.* usually measured at 0.4 x depth) upon which the IFIM habitat model is based. Consequently, when used in the IFIM habitat model, they will tend to underestimate flow requirements of spawning fish. However, the underestimation will be fairly small for the shallow waters preferred by spawning trout because the velocity profile (which is approximated by a power relationship to depth) is compressed in shallow water. However, flow requirements for adult drift-feeding trout are greater than for spawning, or fry rearing.

Given that there is some uncertainty regarding the applicability of the HSC available for trout, the different WUA curves predicted using various HSC may be best interpreted as providing an indication of the range of possible responses of habitat to flow changes. That said, it seems reasonable that suitability criteria should be selected to best represent the habitat needed to maintain a population of the species of interest. Trout populations cannot be expected to be maintained without sufficient feeding habitat, and consequently Hayes & Jowett's (1994) and Roussel *et al.* (1999) criteria might be expected to provide a better estimate of habitat requirements for population maintenance than criteria that are biased toward low flow velocities, by the inclusion of resting trout positions in their development (*e.g.* Raleigh *et al.* 1986). However, suitability criteria developed on markedly different-sized rivers are less likely to be transferable. For comparison, a range of trout habitat suitability criteria were applied in this study (Appendix 1). Conservative flow decisions could safely be made based on the curves predicting highest optimum flow requirements, bearing in mind the size of stream on which the HSC were developed.

The HSC applied for native fish species for this study were mainly based on the work of Jowett & Richardson (2008) (Appendix 1), with the exception of the HSC for longfin eels which were developed based on the work of D. Jellyman and co-workers (Jellyman *et al.* 2003).

Jellyman *et al.* (2003) studied habitat preferences of longfin and shortfin eels in three rivers of varying size (mean flow range 0.04-15 m^3/s) and substrate type. Based on their observations separate sets of HSC were developed for large longfin eels (>300 mm long) and young eels (<300 mm long), with the smaller size class generally preferring shallower, faster water (Appendix 1).

The suitability criteria for shortfin eels <300 mm are based on observations in 58 New Zealand rivers (Jowett & Richardson 2008). Small shortfin eels were recorded from 675 of 2192 locations sampled in these rivers. As with longfin eels these small eels were predominantly found in relatively shallow water, but with lower velocity.

Jowett & Richardson (2008) describe the development of habitat criteria for torrentfish, based on records at 1217 locations in 37 rivers. Torrentfish were usually found in water less than 0.4 m deep and in velocities in excess of 0.5 m/s, but all were found in velocities less than 1.3 m/s.

Jowett & Richardson (2008) recorded observations of koaro in nine rivers. However, the majority of records were from just two rivers (the Onekaka and Ryton rivers), and these records were used to develop the koaro habitat suitability criteria applied here. Koaro were found in association with similarly high velocities to torrentfish, and coarse substrate.

Bluegill bully was the second most abundant species recorded in Jowett & Richardson's (2008) habitat suitability dataset. In total 3253 bluegill bully were recorded at 174 of 764 locations in 15 rivers. About 70% of these fish were found in water between 0.1 to 0.3 m deep, and bluegill bullies had a relatively high velocity preference, second only to torrentfish.

Redfin and upland bullies have quite similar depth and velocity preferences. They are generally found in shallow relatively slow velocity water (Jowett & Richardson 2008). Upland bullies were the most abundant fish recorded in Jowett & Richardson's (2008) dataset, with 3688 fish caught in 523 of 1078 location in 36 rivers. The suitability criteria for Redfin bully are based on fewer observations (564 fish caught from 920 locations in 28 rivers).

None of the native fish HSC applied in this report distinguish between feeding and resting habitat use. Consequently, the same issue of potential slow water bias, as discussed for trout HSC, may also apply. However, the majority of the native fish considered are predominantly benthic feeders, and so probably feed in similar habitat to that which they use for cover. This arguably reduces the importance of water velocity in distinguishing feeding habitat from resting habitat for these fish, compared with drift feeding fish (such as trout and inanga), where the rate of food delivery is directly related to water velocity. In addition, the scale to which physical habitat features are resolved in IFIM habitat modelling (both in the modelling itself and in the measurements on which most of these native fish HSC are based) is probably larger than the scale of habitat use of many small native fishes, in terms of the size of their immediate foraging area. Therefore, the physical habitat conditions modelled as being suitable should be interpreted as being broadly indicative of the type of conditions experienced/preferred by these fish.

Rather than considering individual macroinvertebrate species, the general instream habitat requirements of macroinvertebrates were assessed using Water's (1976) food producing (*i.e.* food for fish) habitat suitability criteria (Appendix 1). These general HSC for benthic macroinvertebrates were developed in the United States of America, but have been widely applied to habitat analyses in New Zealand and Jowett (1992) found that WUA predictions based on them were correlated with trout abundance in New Zealand rivers.

Flow range modelled

The range of flows over which habitat availability can reasonably be modelled is constrained by the flows gauged for the development of rating curves for the survey cross-sections. The further outside the measured flow range that predictions are made the less reliable the predictions are likely to be. Denslinger *et al.* (1998) cite IFIM training documents produced by the U.S. Geological Survey, Biological Resources Division as suggesting that the "hydraulic model [in PHABSIM] can reasonably be extrapolated to a flow equal to 1.5 times the highest calibration flow and 0.6 times the lowest calibration flow. The absolute maximum range for extrapolation is to a flow 2.5 times the highest calibration flow and 0.4 times the lowest calibration flow". These limits are likely to be conservative when applied to RHYHABSIM models, due to improvements made in the way rating curve development is handled in this package (I. Jowett, NIWA, pers. comm.).

The flow range modelled in these habitat analyses was selected to cover the likely range of interest in flow setting decisions, but still provide an indication of the flows at which predicted habitat availability would be optimised. The range extended from above the median flow down to approximately half the mean annual low flow (MALF). The modelled flow range for habitat analysis was well below the upper limit of the suggested absolute maximum range for extrapolation, based on the guidelines outlined above, in both modelled reaches. However, the lower end of the modelled range fell slightly below the suggested lower limit for extrapolation in the Lucy Creek portion of the Upper reach and in the Meads Bridge reach.

5.2. Results and discussion

5.2.1. Response of habitat to flow

The predicted WUA optima for all brown trout HSC modelled occurred above the estimated natural 7-day MALF (Figure 7). The same was true for HSI (Appendix 2). This suggests that any reduction in flow below the natural MALF is likely to lead to a reduction in the amount and average quality of feeding habitat for brown trout in the Lee River. In addition, habitat for invertebrate food production (Waters 1976) was predicted to increase with flow throughout the modelled flow range, so flow reductions would be expected to reduce food supply as well as feeding habitat for trout.

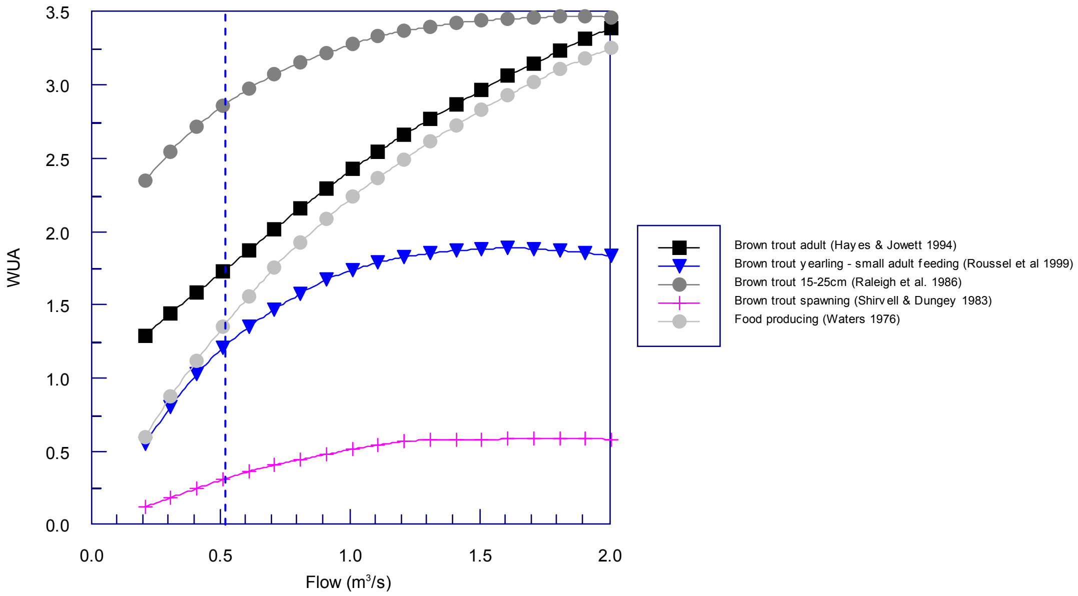

Figure 7. Predicted habitat (WUA) versus flow for trout and macroinvertebrate food producing habitat in the Lee River (Upper and Meads Bridge reaches combined). Blue dashed line denotes natural 7-day MALF.

The same was true for most native fish modelled (Figure 8), although habitat was not predicted to decline as rapidly with reducing flow. Consequently, a given level of flow reduction from the MALF would be expected to produce a lesser degree of habitat reduction for the native fish modelled than for trout.

For upland bullies, WUA was predicted to increase with reducing flow close to the MALF. This suggests that moderate reductions in flow from the naturalised 7-day MALF should be beneficial for this species, at least with respect to habitat availability.

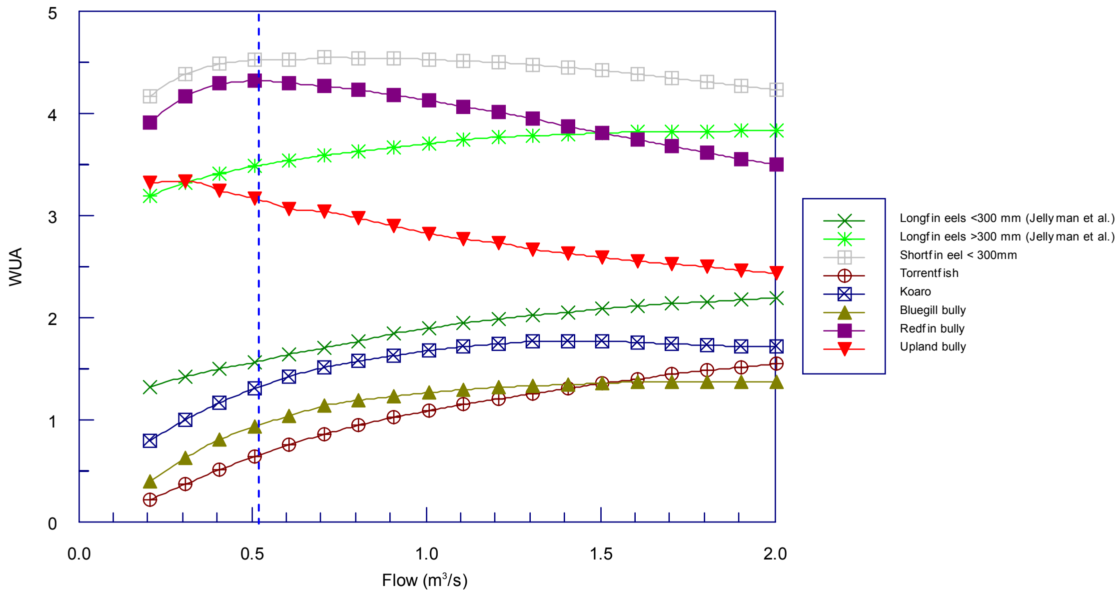

Figure 8. Predicted habitat (WUA) versus flow for native fish habitat in the Lee River (Upper and Meads Bridge reaches combined). Blue dashed line denotes natural 7-day MALF.

5.2.2. *Interpretation of WUA curves for flow management*

Ecological relevance of the MALF

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

The reason why the MALF is a potential limiting factor, for trout populations, is that it is the most commonly used flow statistic that is indicative of the average annual minimum living space for adult trout. Trout populations respond to annual limiting events because their cohorts (year classes) are annual (*i.e.* they reproduce only once per year). This contrasts with aquatic invertebrates, which in New Zealand generally have asynchronous lifecycles (*i.e.* a range of different life stages are likely to be present at any given time) and may also have more than one cohort per year so their populations respond to more frequent limiting events (*e.g.* floods or low flows that occur over the time-scale of months). Other flow statistics that define, or are closely correlated with, average annual minimum flows should be similarly relevant as the MALF to adult trout abundance.

Jowett's research provides empirical and conceptual justification for the validity of WUA as a habitat index for trout populations in New Zealand rivers. The insights gained from this research can also provide a basis for identifying hydrological statistics that are ecologically relevant to trout populations. It seems reasonable that the MALF should be similarly relevant to native fish species with generation cycles longer than one year, at least in situations where habitat declines toward the MALF. If the minimum flow restricts habitat for any species, there is potential for a detrimental effect on that population if abstraction draws flow below the MALF.

These insights have led to a recent move toward interpreting WUA curves in conjunction with flow statistics (notably the MALF) when making decisions on minimum flows (Jowett & Hayes 2004). It has been suggested that if the WUA optimum should occur at flows above the MALF, then habitat availability will be limited by the MALF. In this case, flow decisions should be made so as to preserve a proportion of the habitat (*i.e.* WUA) available at the MALF (Figure 9a), in order to cater for the needs of both instream values and out-of-stream water uses. In the case where predicted optimum WUA occurs below the MALF, then flows should be managed to maintain a proportion of the habitat available at optimum WUA (Figure 9b).

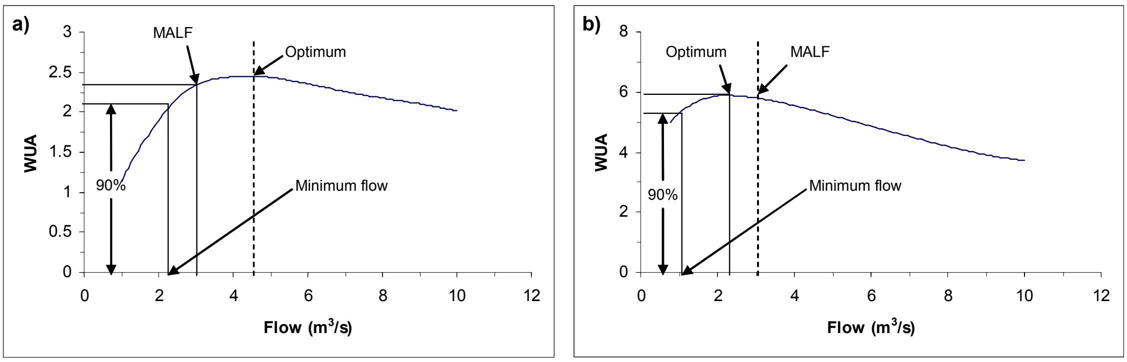

Figure 9. Derivation of minimum flow based on retention of a proportion (90% in this case) of available habitat (WUA) at a) the MALF, or b) the habitat optimum, whichever occurs at the lower flow, as recommended by Jowett & Hayes (2004).

Reconciling flow requirements of multiple instream values

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

Trout are recognised as being among the most flow-demanding fish in New Zealand rivers. As discussed by Hay & Young (2005b) in the Phase 1 study, brown trout support a recreational fishery in the Lee River, although difficulty with access tends to restrict fishing opportunities, and the river does not attract high numbers of anglers. The Lee River ranked 53rd, in terms of angler days, among 88 water bodies in Fish & Game's Nelson Marlborough Region, in the latest national angler survey (Unwin 2009), with 50 angler days during the 2007-2008 season. Angler usage during that survey was lower than during either of the previous two survey periods (80 angler days during the 2001-2002 season and 130 angler days during the 1994-1995 season).

Of those native fish species recorded from the Lee River (Table 5), one is of particular conservation concern based on the latest Department of Conservation threat classification listings (Hitchmough *et al.* 2007). Longfin eel is listed as "declining" (adapted to the revised threat classification system of Townsend *et al.* 2008). Although this species is not generally considered to be particularly flow demanding, habitat suitability criteria for small eels indicate that they prefer relatively fast and shallow riffle habitat (Jellyman *et al.* 2003).

Koura is also listed as "declining". However, there are no habitat suitability criteria available to model habitat changes with flow for this species of crustacean.

Among the native fish, torrentfish and bluegill bullies are the species with the highest flow demands. However, neither of these species is considered to be threatened, in fact both are relatively widespread and common, and neither supports a fishery. Consequently, although their flow demands are relatively high, they are of relatively low value (existence value) compared to trout and eels.

Koaro have slightly lower flow requirements than torrentfish and bluegill bullies. Although this galaxiid species is still widespread and not considered threatened, its returning marine migratory juveniles do contribute to the "whitebait" fishery. Koaro are generally the second largest contributor numerically to the whitebait catch, after inanga (McDowall 2000), and in some areas they comprise a major proportion of the catch (particularly in some West Coast rivers; McDowall 1965).

This analysis suggests that trout, small eels, or koaro are possible candidates for critical value status in the Lee River, and since the flow requirements of trout are higher than those of small eels or koaro, trout would be the most appropriate critical value. Providing for the flow needs of trout will, arguably, provide for the flow needs of less flow demanding species, because these will be able to utilise slower or shallower habitat along the river margins, or in riffles or pools. The habitat requirements of yearling to adult brown trout for feeding are arguably the most pertinent to minimum flow setting for this river.

Habitat retention levels

Finally, the decision remains as to what level of habitat availability should be maintained. The level of habitat retention is arbitrary, and scientific knowledge of the response of river ecosystems, and fish populations in particular, is insufficient to identify levels of habitat below which ecological impacts will occur. A carefully designed and well funded monitoring programme might detect effects of a 50% reduction in habitat on fish populations but is unlikely to detect effects of a 10% reduction in habitat – due mainly to the large natural spatial and temporal variability typical of fish populations. It is uncertain whether any effects of a 20-30% reduction in habitat on fish populations would be detectable.

Jowett & Hayes (2004) recognise that, in practice, the choice of a habitat retention level is based more on risk management than ecological science. The risk of ecological impact increases as habitat is reduced. When instream resource values are factored into the decision-making process, then the greater the resource value the less risk is acceptable. With this in mind, Jowett & Hayes (2004) suggest that water managers could consider varying the percent habitat retention level, depending on the value of instream and out-of-stream resources (*i.e.* highly valued instream resources warrant a higher level of habitat retention than low valued instream resources). This concept is consistent with conservative flow decisions in water conservation orders, which generally allow for only relatively small reductions in flow or habitat. Table 6 shows how Jowett & Hayes (2004) envisage that percentage habitat retention could be varied to take account of variation in instream values.

Table 6. Suggested significance ranking (from highest (1) to lowest (5)) of critical values and levels of habitat retention.

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

Table taken from Jowett & Hayes (2004)

5.2.3. *Proposed minimum flows for the Lee River*

On the basis of the rationale outlined above, the minimum flows proposed here for the Lee River are based on retention of a proportion of the predicted habitat (WUA) available at the MALF for yearling to adult brown trout (since their WUA optima occurred at flows above the MALF). Minimum flows based on three levels of habitat retention are presented (Table 7), with the intention that these might provide options for negotiations on the relative values of instream and out-of-stream water use. The natural 7-day MALF (0.51 m^3/s) is proposed as the environmental benchmark minimum flow for the Lee River below the proposed dam. A minimum flow of 0.32 m^3/s would retain 70% of the yearling to adult brown trout habitat available at the natural MALF (Table 7). An intermediate option is a minimum flow of 0.38 m^3/s, which would retain 80% of the habitat available at the natural MALF for yearling to adult brown trout (Table 7).

The flow requirements for trout spawning are slightly higher than those for yearling to adult fish in these reaches (Table 7). For this reason a higher minimum flow may be warranted during the winter spawning and incubation season (say May to November, inclusive).

For comparison Table 7 also contains minimum flows that would be required to provide similar levels of habitat retention for the other species modelled. In keeping with the suggested habitat retention levels in Table 6 the 80% and 70% habitat retention levels were applied to all those species identified as having specific conservation or fisheries values, while slightly lower retention levels (70% and 60% retention) were applied to those species with purely existence value.

As expected, the minimum flows based on habitat retention for the native fish modelled were consistently lower than for brown trout. The fast water specialist native fish (torrentfish, bluegill bullies and koaro) had similar minimum flow requirements to those based on trout habitat (WUA) retention. This supports the contention that setting a minimum flow to protect trout habitat availability should also accommodate the minimum flow requirements of native fish.

We understand that the natural 7-day MALF (0.51 m^3/s) has been adopted as the environmental benchmark minimum flow for the Lee River below the proposed dam and incorporated in design and operating regime determination.

Table 7. Flows at predicted WUA optima and flows predicted to retain 80% and 70%, or 70% and 60%, of the WUA at the MALF or the flow at the WUA optimum (whichever is lowest) for all species and life-stages modelled in the Lee River. The proposed minimum flows are highlighted in bold.

Habitat Suitability Criteria	MALF (m^3/s) (*i.e.* 100% habitat retention at the MALF)	Flow at WUA Optimum (m^3/s)	Flow that retains 80% of WUA at MALF or the WUA optimum flow (m^3/s)	Flow that retains 70% of WUA at MALF or the WUA optimum flow (m^3/s)
Brown trout adult (Hayes & Jowett 1994)	**0.51**	2.00	0.27	**
Brown trout yearling - small adult feeding (Roussel *et al.* 1999)		1.58	**0.38**	**0.32**
Brown trout 15-25cm (Raleigh *et al.* 1986)		1.83	**	**
Brown trout spawning (Shirvell & Dungey 1983)		1.67	**0.41**	**0.35**
Longfin eels <300 mm (Jellyman *et al.* 2003)		2.00	**	**
Longfin eels >300 mm (Jellyman *et al.* 2003)		1.88	**	**
Shortfin eel < 300 mm (Jowett & Richardson 2008)		0.70	**	**
Koaro (Jowett & Richardson 2008)		1.38	0.33	0.26
			Flow that retains 70% of WUA at MALF or the WUA optimum flow (m^3/s)	Flow that retains 60% of WUA at MALF or the WUA optimum flow (m^3/s)
Torrentfish (Jowett & Richardson 2008)		2.00	0.36	0.32
Bluegill bully (Jowett & Richardson 2008)		1.83	0.32	0.27
Redfin bully (Jowett & Richardson 2008)		0.47	**	**
Upland bully (Jowett & Richardson 2008)		0.25	**	**

** below modelled range

5.2.4. Flow variability and the minimum flow

It is important that maintenance of flow variability be considered in conjunction with setting minimum flows, to maintain channel and riparian structure, control periphyton, and sustain invertebrate productivity and fish feeding opportunities. Along with the magnitude of the minimum flow, increasing the frequency and duration of occurrence of the minimum flow is likely to have ecological effects.

Perhaps the most obvious potential ecological effect of prolonged low flow, due to abstraction, is proliferation of periphyton to nuisance levels, which may be mitigated by flushing flows. But impacts are likely to extend to higher trophic levels (*i.e.* invertebrates and fish) as well. In the past, minimum flows in New Zealand have generally been set under the assumption that abstraction is unlikely to have a significant impact on the hydrograph other than low flows (except when large dams with substantial storage capacity are involved). However, nowadays there is more demand on water and moderate to large scale water abstraction may well significantly alter other features of flow regimes, although this is more likely with water abstraction from relatively small streams. These changes generally do not affect flood and flushing flows but may affect the availability of invertebrate food resources for fish and birds by temporarily reducing invertebrate habitat, with associated reduction in invertebrate production. Generally, optimal invertebrate habitat occurs at higher flows than optimal fish habitat and because they have high rates of colonisation, invertebrates can make productive use of extended flow recessions. For instance, they may take as little as 15-30 days to fully colonise previously dry channels (or margins) (Sagar 1983). For this reason the median flow can be thought of as providing an approximation of the habitat conditions experienced, and able to be utilised, by benthic invertebrates most of the time (Jowett 1992).

In comparison, the minimum flow can be viewed as providing essentially a habitat refuge for fish during periods of low flow. It should not be viewed as providing adequate habitat to support fish populations over the long-term if flow is consistently held at the minimum, because food supply for fish is likely to be reduced. Setting a minimum flow at or below the MALF with no safeguards for maintenance of flow variability has been likened to a doctor prescribing a patient's worst state of health as a life-time condition. The aim in setting the minimum flow is to provide enough suitable habitat for fish to survive in, hopefully fairly comfortably, for a relatively short period before flow increases again. NIWA research in the Waipara River, where fish habitat is limited at low flow, showed that the detrimental effect on fish numbers increased with the duration and decreasing magnitude of the minimum flow (Jowett & Hayes 2004).

Maintenance of invertebrate production (which fish depend on for food) is arguably more dependent on ensuring that the median flow is not substantially reduced by abstraction, than on the minimum flow *per se*.

In the current study, flows in the Lee below the dam are likely to be above the minimum flow much of the time, since the aim of the overall project is to augment flow lower in the catchment. However, flow may be close to the minimum during periods when the reservoir is refilling (*i.e.* is not spilling) and flow in the lower catchment is being held above minimum levels by inflow from other parts of the catchment (*e.g.* the Roding and Wairoa rivers).

To assess the effect of the proposed augmentation regime on habitat availability in the Lee River, the habitat (WUA) at monthly median flows under the natural flow regime has been compared with that under the augmented flow regime scheme. This comparison was based on monthly median flows, calculated from data provided by Tonkin & Taylor simulating natural daily average flows and preliminary predicted daily average outflows from the dam for the hydrological years July 1957 to June 2007.

As expected, this analysis showed that the main effect of the augmentation scheme is to increase flow during the summer irrigation, and low flow, season (Figure 10). This was predicted to peak with a 57% increase in the monthly median flow in February. The increase in monthly median flows during summer result in a predicted increase in habitat availability (WUA) for the majority of species modelled. The exceptions are redfin and upland bullies, and shortfin eel <300 mm in length, for which the increase in flow is predicted to cause a small (<10%) reduction in habitat availability. From April to November (inclusive) there is no more than a 3% change in predicted habitat at the monthly median flows under the natural versus augmented flow regime for any of the species modelled.

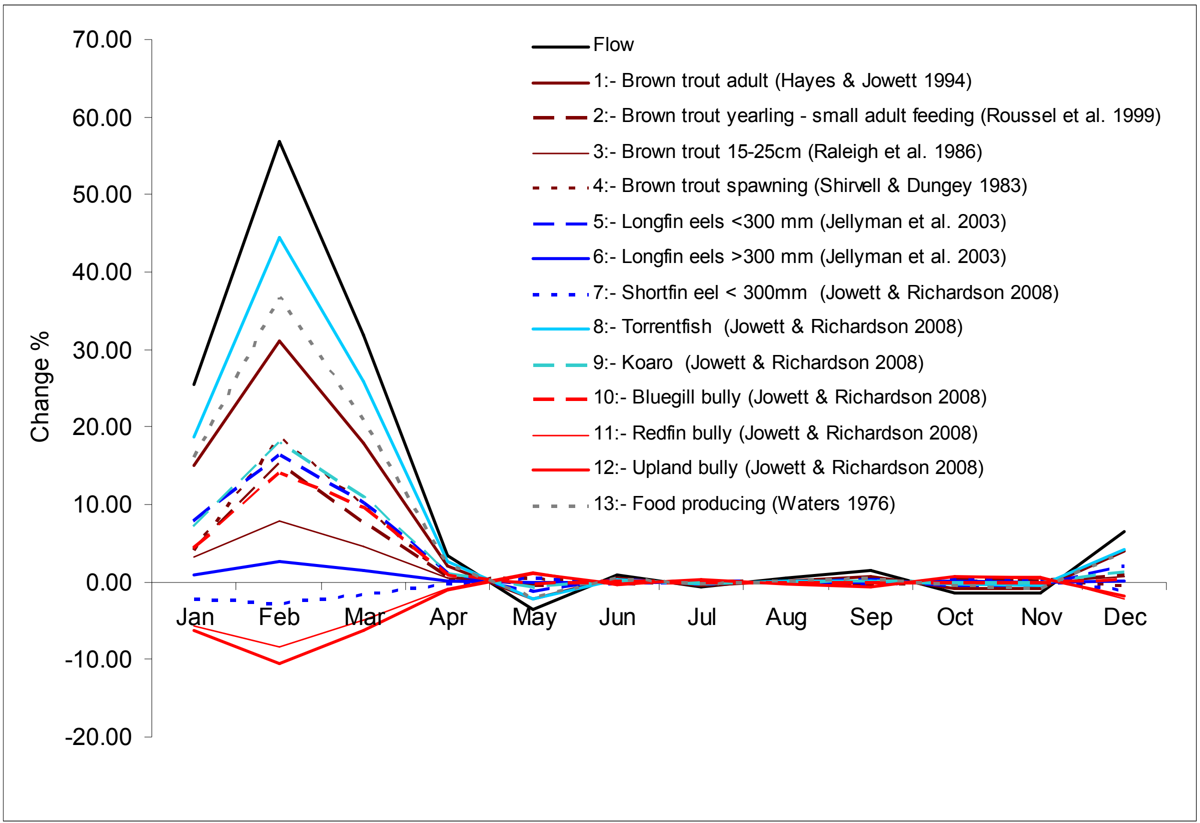

Figure 10. Predicted changes in habitat available (WUA) at monthly median flows under augmented flow regime compared with that under the natural flow regime for the species modelled in the Lee River.

This analysis provides an indication of the changes in monthly median flow and habitat that can be expected in a typical year. However, in years with extremely low flows during summer the changes in flow and habitat relative to the natural situation would be greater, both through the summer irrigation system and the autumn to early winter period as the dam is refilled. This is illustrated by Figure 11, which shows the same analysis undertaken for 1973 – the year with the driest February during the period of record (1957-2007). This shows essentially the same pattern of increased flow and habitat for most species during summer, and a period of decreased flow and habitat for most species during the early winter, as the reservoir refills. The difference is that the magnitude of change is much larger, with close to a 500% increase in the median flow for February and for median torrentfish habitat in that month, and about a 60% reduction in monthly median flow and torrentfish habitat during May and June. Monthly median brown trout spawning habitat was also predicted to be reduced by 50% during these months, while habitat for yearling and adult brown trout was predicted to be reduced by about

39%. However, it should be borne in mind that this year represents an extreme case, as it had the driest February over a 50 year period.

Again there was very little change in flow or habitat during the majority of winter and spring (July to November in this case), when the reservoir would mainly be full and spilling. By the same token, in years with particularly wet summers the changes in flow and habitat will be small.

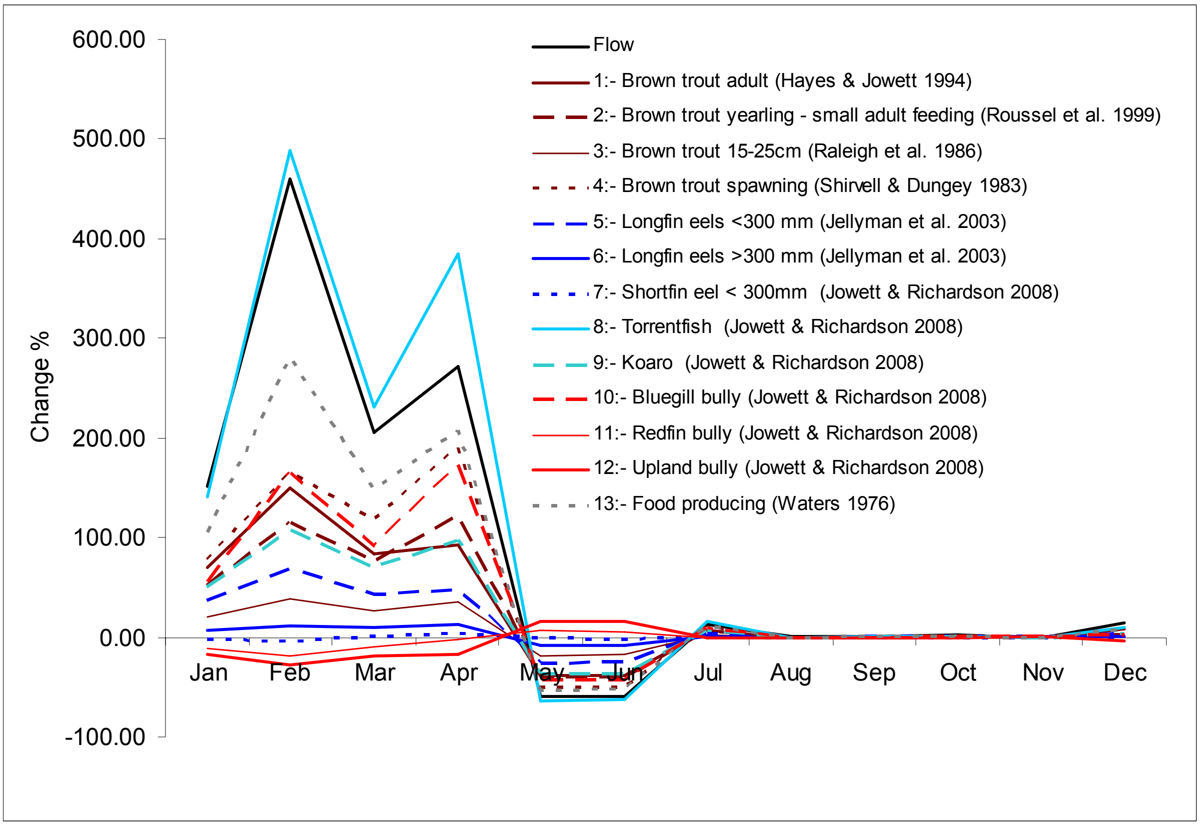

Figure 11. Predicted changes in habitat available (WUA) at monthly median flows under augmented flow regime compared with that under the natural flow regime during 1973 (the year with driest February in the period of record, 1957-2007) for the species modelled in the Lee River.

It is possible that hydro-power generation will be incorporated into the scheme design. Synthesised flow regimes downstream of the dam based on without-hydro and base-load hydro scenarios (supplied by Tonkin & Taylor Ltd) indicate very minor effects on key ecologically relevant flow statistics. For example, the MALF at Wairoa at Irvines is predicted to be 2.88 m^3/s with hydro and 2.86 m^3/s without hydro. Similarly, median flows at the same site are predicted to be 7.6 m^3/s with hydro and 7.2 m^3/s without hydro. Such small differences are predicted to have a negligible effect on habitat availability. However, any potential effects of hydro-peaking on habitat availability downstream were beyond the scope of this study.

Therefore, if hydro-peaking were to be considered during a later hydro-power optimisation study, then the specific effects on habitat would need to be assessed.

5.2.5. *Weighing up the costs and benefits of the scheme on instream habitat*

The creation of the reservoir, restriction of fish passage and changes in the flow regime downstream of the proposed dam potentially have positive and negative effects on instream habitat availability and aquatic life. For example, the increased minimum flows downstream of the proposed reservoir are predicted to result in a 25% increase in the number of adult trout in the lower Waimea River (from 15/km to 19/km) using a model from Jowett (1992) (See Hay & Young 2005a for more detail). On the other hand, redfin bullies are unlikely to negotiate the fish pass over the dam and access any habitat upstream.

To assess the net effect associated with the proposed water storage scheme we predicted habitat availability (WUA) for the range of species present in different reaches of the catchment and multiplied this by the length of river affected. For this analysis the river was divided into five sections:

1. Wairoa/Waimea River from Irvines recorder to the sea,
2. The Lee River from the Roding River confluence to Irvines Recorder,
3. The Lee River from the Dam to the Roding confluence,
4. The reach of the Lee River within the dam footprint, and
5. The Lee River upstream of the dam footprint.

Habitat availability for the Wairoa/Waimea River from Irvines Recorder to the sea, the Lee River from the Dam to the Roding confluence, and the reach of the Lee River within the dam footprint and upstream was estimated from the IFIM surveys that have been conducted in or near these reaches (as reported above and in Hay & Young 2005a). No information on habitat change with flow is available for the reach of the Lee River between the Roding confluence and Irvines Recorder so it was assumed that there would be no change in habitat availability in this reach as a result of the scheme. This is likely to be a conservative approach for most species, since flow will generally be increased in this reach with a consequent increase in habitat availability. For the reach of the Lee Catchment above the reservoir footprint we only accounted for habitat availability in streams of 3rd order or greater.

We used the change in median flow resulting from the scheme to infer changes in macroinvertebrate habitat availability, and changes in 7-day MALF resulting from the scheme to infer changes in habitat for fish. The rationale for this difference is that macroinvertebrates have relatively short life cycles therefore their abundance is likely to be controlled by the amount of habitat available most of the time (as indicated by median flow) (see discussion in Section 5.2.4 above). However, for fish we have assumed that the minimum flow experienced every couple of years (MALF) is the bottleneck through which fish populations must pass (see discussion in Section 5.2.2 above). In other words, fish are unable to capitalise on short-term increases in habitat availability in the same way that invertebrates can.

Based on this rationale we added up the positive and negative effects of the scheme throughout the catchment. This included taking account of habitat losses upstream of the dam for species that are unlikely to pass the dam, but assuming that habitat created in the reservoir compensates for loss of riverine habitat within the reservoir footprint for those species that are likely to be present above the dam. Approximately 80,000 m^2 of riverine habitat will be inundated by the reservoir (5.4 km of river channel x 14.7 m average width), compared with approximately 650,000 m^2 of lake habitat created within the reservoir.

For migratory (diadromous) fish species we also weighted the habitat value in each reach by the predicted fish density for each river section based on their elevation, given the typical reduction in abundance of these species with distance from the coast (Richardson & Jowett 1996).

The results of this analysis are shown in Table 8. Overall we predict a positive net effect for adult trout, small trout, eels, torrentfish, koaro, upland bully, and food producing habitat, but a net negative effect for yearling trout and redfin bullies. Bluegill bullies are expected to be affected negatively based on the raw numbers and affected positively based on the fish density weighted numbers - *i.e.* improvements in habitat availability near the coast are more influential than loss of habitat further inland.

In summary, most species are predicted to benefit from the water augmentation scheme, primarily in response to the increased minimum flows in the lower catchment. The main exception is redfin bullies, which tend to like slow shallow water and thus will not benefit from enhanced minimum flows in the lower reaches of the river. Redfin bullies will also be unlikely to negotiate the fish pass and occupy habitat above the dam.

Table 8. Predicted changes in net instream habitat availability (WUA) for a range of species and life stages throughout the Waimea Catchment that are associated with the proposed water storage scheme.

Species/Life Stage	Net WUA change	Net WUA change weighted by fish density
Brown trout adult (Hayes & Jowett 1994)	12.1	60.4
Brown trout yearling (Roussel et al 1999)	2.1	10.5
Brown trout 15-25cm (Raleigh et al. 1986)	8.3	41.6
Longfin eels <300 mm (Jellyman et al.)	7.4	231.5
Longfin eels >300 mm (Jellyman et al.)	5.9	185.5
Shortfin eel < 300mm (Jowett & Richardson 2008)	5.9	195.5
Torrentfish (Jowett & Richardson 2008)	8.0	603.9
Koaro (Jowett & Richardson 2008)	1.3	20.7
Bluegill bully (Jowett & Richardson 2008)	-4.7	353.0
Redfin bully (Jowett & Richardson 2008)	-85.3	-316.3
Upland bully (Jowett & Richardson 2008)	1.2	6.1
Food producing (Waters 1976)	0.5	

6. FLUSHING FLOWS

During prolonged periods of stable, low flow periphyton and fine sediments can accumulate to excessive levels in the streambed. In RHYHABSIM it is possible to predict the flows required to flush fine sediment and periphyton from the streambed (Jowett *et al.* 2008). The RHYHABSIM hydraulic model described above (Section 5.1) was used to model flushing flows.

A flushing flow analysis involves predicting the area of the streambed that will be disturbed by given flows, and the particle sizes that will be moved in suspension and as bed-load. The calculation of the amount of disturbance is based on shear stress. Using data from a small gravel bed stream, Milhous (1998) showed that surface sediments were flushed when the dimensionless bed shear stress exceeded 0.021 and that the armour layer was disturbed when the stress exceeded 0.035. These values are used in RHYHABSIM to calculate the area of the stream bed that is flushed by a given flow.

The flushing flow analysis for the Lee River focused on the baseflow channel (*i.e.* the channel that would be wetted under a relatively low flow, "baseflow"), since periphyton proliferation is most likely to occur under relatively prolonged periods of stable low flow. For this analysis the baseflow was taken to be 0.5 m^3/s (close to the natural 7-day MALF) and the proportion of the baseflow channel that would be flushed was assessed over a range of flows.

Within RHYHABSIM shear velocity ($v*$) was calculated for each point in the modelled reaches, based on the mean column velocity (V), acceleration due to gravity (g), hydraulic radius (R), and Manning's n.

$$v* = \sqrt{g} \cdot V \cdot n / R^{1/6}$$

Where Manning's n was estimated based on the substrate composition recorded at each measurement point on each cross-section. And dimensionless shear stress was then calculated within RHYHABSIM as:

$$\text{Dimensionless Shear Stress} = = \sqrt{v*}\ /g/(SG\text{-}1)/\text{substrate size}$$

Where SG is the specific gravity of the substrate (taken to be 2.65) and substrate size is the d85 substrate size estimated from percentage substrate composition at each point (I Jowett, RHYHABSIM developer, pers. comm.).

For this modelling, the velocity distribution factors (VDFs) calculated from the velocity distribution at the survey flow were not applied; rather the velocity in each cell of each cross-section was calculated according to the hydraulic conveyance of that cell. This is because transverse velocity variation generally tends to decrease at high flows, as the objects and constrictions that drive the velocity distribution observed during low flows are drowned out.

RHYHABSIM also predicts the maximum suspended sediment and maximum bedload particle size that will be mobilised at each flow based on formulae presented by Milhous (1998).

Although sediment transport occurs at practically all flows, as flow increases the amount and size of sediment that is transported also increases. Some areas of the stream bed tend to resist movement more than others, but the area of stream bed that is disturbed by high flows gradually increases as the flow increases (Jowett *et al.* 2008). Surface flushing flows remove the fine sediments from the surface layer, leaving the armour layer largely intact. Periphyton can also be removed by the abrasive action of fine sediments moving over the surface (analogous to sandblasting).

Flushing flow analysis indicated that a flow of about 3 m^3/s is necessary to initiate flushing of fine sediment and periphyton from the bed of the Lee River below the proposed dam (Figure 12). However, flows of 4.5-5 m^3/s were predicted to be required before appreciable effects of flushing would begin to occur. That flow range is predicted to suspend sediment up to the size of coarse sand to fine gravel, but only over a relatively small proportion of the river bed (~5% of the baseflow channel). It is also predicted to mobilise substrate up to ~20 mm gravels as bedload (*i.e.* moving by rolling or sliding along the bed) in places.

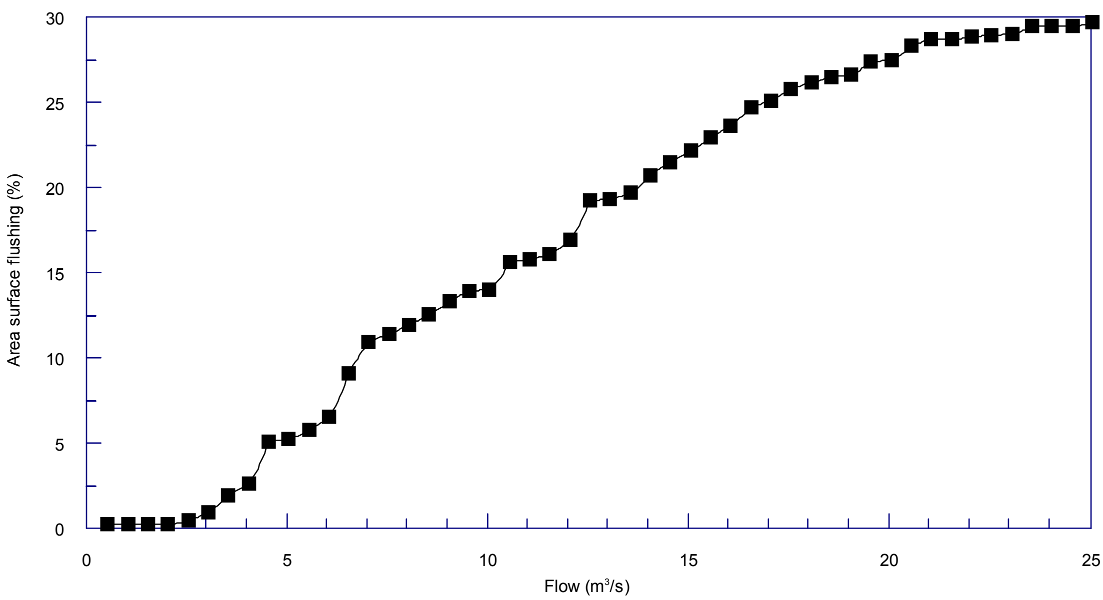

Figure 12. Predicted surface sediment flushing as a function of flow for the Lee River, expressed as a percentage of the area of wetted bed at a baseflow of 0.5 m^3/s.

The area flushed is predicted to increase reasonably steadily, to reach approximately 25% of the baseflow channel at 16.5 m^3/s, when the average dimensionless shear stress over the modelled reach would be 0.021. At this flow gravel up to about 11 mm diameter is predicted to be carried in suspension, and gravel as coarse as 57 mm would be carried as bedload.

Beyond this flow the percentage of the baseflow channel with sufficient shear stress to cause surface flushing continues to rise steadily.

There is no obvious break point over the modelled flow range where increasing flow further begins to provide diminishing returns in terms of the proportion of the bed flushed. However, flows lower than about 5 m^3/s are not likely to cause any appreciable flushing. A flow of 5 m^3/s is predicted to cause some surface flushing of fine sediments, and should also flush periphyton from at least some of the bed, especially given the scouring effect of the fine sediment that would be moving in suspension. Clausen & Biggs (1997) reported a strong negative relationship between periphyton (measured using chlorophyll *a* concentration) and the frequency of floods >3 times the median flow. This provided a commonly used rule of thumb for periphyton flushing flows of three times the median flow. In this case the natural median flow for the Lee River is 1.71 m^3/s, and the post-scheme median flow is predicted to be only slightly higher at 1.76 m^3/s; so this rule of thumb would suggest a flushing flow of 3 x 1.71 m^3/s = 5.13 m^3/s, or 3 x 1.76 m^3/s = 5.28 m^3/s, which closely corresponds with the minimum flow predicted to cause appreciable surface sediment flushing with the modelling.

An alternative rule of thumb is a flow of 6-8 times the preceding baseflow is required to reduce periphyton biomass (Biggs & Close 1989; Biggs 2000). Given a baseflow of 0.5 m^3/s this would give a flushing flow of 3-4 m^3/s. Therefore, the 5 m^3/s flushing flow suggested above should be conservative.

Higher flows, with stronger flushing potential, will continue to occur with similar frequency to the present natural flow regime under the proposed operating regime. However, at those times when the reservoir has been drawn down some freshes will be captured by the dam. During these periods the flow below the dam may be held at a relatively stable low level for some time, which may allow periphyton to develop to nuisance levels. Having the ability to release flushing flows in the order of 5 m^3/s from the base of the dam would provide the potential to mitigate this issue.

7. FISH PASSAGE MITIGATION

Fifteen species of fish and one crustacean have been recorded from the Waimea catchment (Hay & Young 2005b). Fish distributions ascertained from these records indicate that 8 species of fish and a crustacean (freshwater crayfish) were recorded from the vicinity of the proposed storage reservoir in the Lee River (Table 9). Both eel species, koaro, torrentfish, and redfin and bluegill bullies have life cycles that require access to and from the sea (Table 9).

Table 9. Fish species recorded from the vicinity of the proposed storage reservoir in the Lee River

Common name	Scientific name	Life cycle
Longfin eel	*Anguilla dieffenbachii*	Migratory
Shortfin eel	*Anguilla australis*	Migratory
Koaro	*Galaxias brevipinnis*	Migratory
Brown trout	*Salmo trutta*	Freshwater
Torrentfish	*Cheimarrichthys fosteri*	Migratory
Redfin bully	*Gobiomorphus huttoni*	Migratory
Bluegill bully	*Gobiomorphus hubbsi*	Migratory
Upland bully	*Gobiomorphus breviceps*	Freshwater
Freshwater crayfish	*Paranephrops planifrons*	Freshwater

Given the height of the proposed dam (approximately 52 m) and the relatively low status of the trout fishery in the Lee River, it is considered that mitigation of fish passage issues associated with the dam is necessary and practical for only the strongest of migrants such as elvers and young koaro. Both these species will attempt to scale structures such as the proposed dam provided they have an uninterrupted wet surface that leads from the downstream base of the dam to permanent water in the reservoir. Most upstream migratory attempts could be expected to occur from November to February.

As part of the Phase 1 study, Hay *et al.* (2006) suggested that there are two options to facilitate passage for these strong migrants; an uninterrupted wet surface that is incorporated in the spillway design enabling passage at any time for fish to migrate. Alternatively, an attraction flow leading from a trap could be installed so that migratory species could be caught and manually transferred upstream. Of the two options, a functioning pass is preferred since it is less expensive to maintain and would be permanently in place to provide access at any time fish attempt to migrate. A trap, on the other hand, requires maintenance and unless operated continually could miss some migrations. However, traps do have the benefit of “feel good” about them in that people are able to see and count what is transferred regardless of the biological significance of the transfer (Hay *et al.* 2006).

Assuming access is provided past the dam for koaro and eels, some consideration then needs to be given to providing their return access downstream. Koaro require downstream access after spawning in autumn when their larvae passively migrate downstream during a fresh. Consequently, they will be naturally entrained, either via augmentation releases or spilling. The majority of downstream movement will occur during freshes, when the dam is likely to be spilling and koaro larvae will be carried downstream in the spillway flow. Alternatively, those larvae that are not carried past the dam may remain and rear in the reservoir. Natural mortality of koaro larvae as they are shunted downstream is unknown. However Coutant &Whitney (2000) report that survival of planktonic fish through the extreme conditions associated with hydro-power turbines is high. On this basis survival is also likely to be high for larval fish passing downstream in the spillway flow, and there is likely to be little advantage to downstream migrating koaro in attempting to provide an alternative downstream pathway.

Eels present a slightly different problem because they migrate downstream as mature (and often very large) adults. Depending on how they exit the dam, they can suffer damage and mortality, though in the absence of turbines or screens, this may be less of an issue. Eel downstream migration occurs during autumn freshes.

There are likely to be few options for enhancing downstream migration of koaro larvae or adult eels other than releasing some flow from the reservoir during autumn freshes when the strongest likelihood of these fish seeking downstream access will occur. Allowing release of water through the spillway rather than the intake may allow fish, particularly eels, a better chance of locating the spillway exit. However, many natural autumn freshes may be 'captured' within the reservoir as water levels recover following flow augmentation over the summer. Spilling at the appropriate time of year is not likely to occur during dry years. Lake and reservoir populations of eels are often restricted to downstream migration during years when there are sufficient freshes to bring about spillway flow and allow access out. As a contingency for successive dry years that produce no spilling during autumn, the only feasible option to facilitate downstream migration would be to trap migrants and manually transfer them downstream over the dam wall.

Iterative discussions have been held between the dam designer (Tonkin & Taylor), Cawthron, Fish & Game NZ and DOC. Several initial design options have been ruled out during this process due to design difficulties and/or their incompatibility with fish passage.

One, recently discussed, option incorporated a fish passage channel into the spillway to keep the flows concentrated in one part of the spillway and allowing fish passage up the wetted margins. A continuous flow was to be artificially provided down both sides of the dam crest to ensure year round passage. A slot passing through the flip bucket would have provided access from the plunge pool to the spillway fish pass. While this concept provided some relatively simple build solutions and showed promise for fish passage, it would have required a second channel from the bottom of the spillway across to the main reservoir outflow (*i.e.* the augmentation flow release outlet), where upstream migrating fish would naturally be attracted and accumulate. The design also only catered for the reservoir being used for irrigation purposes and posed further issues around re-routing the fish pass if hydro-electric use were to be incorporated in the project. A further potential issue of this design was the risk of damage that the fish pass slot in the spillway flip bucket would have posed for downstream adult eel migrants. A simpler fish passage option was arrived at that will work for any of the uses that are being considered for the Lee dam.

For the purpose of providing access for upstream migrating fish, it is proposed that a small channel fed by water pumped from the reservoir be located on the faces of the dam. The channel will take the form of a rock-filled nature-like design, winding its way down and past both sides of the dam. The outlet to the fish pass channel will be located at the downstream side of the augmentation flow release outlet and on the reservoir side will continue to a level that allows fish pass connectivity at a range of reservoir levels. Rodent control around the channel would be wise to limit predation of migrating fish.

It is more difficult to predict the likely effects of the dam on downstream migrating adult eels. As has been explained earlier, adult eels typically migrate downstream during autumn flood events. These migrations would only be successful when the spillway was operating and at such times the flip bucket has the potential to cause abrasion damage and/or mortality to these fish. However, these migrations are likely to occur near the peak of flood spills, when sufficient depth of water down the spillway and out over the flip bucket should prevent abrasion damage – depending on where in the water column the eels are transported down the spillway. With the removal of the fish pass slot concept there will now be a smooth unobstructed pathway for water to be directed down the spillway and across the flip bucket.

However, a further danger for eels transported downstream in this manner could arise from impact injury as they land in the plunge pool. There appears to be sufficient depth in the plunge pool design and a lack of obstructions in the plunge pool ought to avoid too much of a danger in this regard. However, since there is some uncertainty regarding the likelihood of injury or mortality among downstream migrating eels passed over the spillway, this issue should be the subject of monitoring when the dam is operational. If it is found that injury or mortality rates are high, alternative approaches, such as trapping and manually transferring migrants downstream, may need to be implemented.

A fish screen at the intake would only be required to protect downstream migrating eels AND if there are turbines as part of the outlet. Protection for downstream migrating eels would mainly be necessary at peak flood flows. This migration is surface oriented and would most likely be drawn towards the spill flow as a means of exit from the reservoir. However if screening is required, a mesh size of 20 mm and an approach velocity around 0.3 m/s or perhaps a bit higher is recommended since the focus will be on mature eels, not weak swimming juveniles.

There may be some issues with fish passage during the construction period. For example, the diversion culvert may be a barrier for some species, but given that it will be large enough to cope with moderate sized floods, passage for strong migrants should still be possible. Ineffective sediment management during construction is likely to pose a bigger risk to fish passage and river health downstream. Sediment control measures during construction will need to be a high priority.

8. RESERVOIR MANAGEMENT

8.1. Lake fishery

As discussed by Hay *et al.* (2006) in the Phase 1 study, the fish community upstream of the dam is likely to comprise brown trout, upland bullies and freshwater crayfish, and with provision of fish passage past the dam for climbing species, is likely to also include longfin and shortfin eels, and koaro.

A self-sustaining trout fishery in the reservoir will be reliant on adequate spawning and rearing habitat in the upper catchment, and the size of the fishery supported will depend on the productivity of the lake. The availability of spawning habitat in the tributaries of the upper Lee catchment is largely unknown. The section of Waterfall Creek below the waterfall currently contains gravels that appear suitable for spawning and supports juvenile trout (J. Hay, pers. obs.). However, the majority of the section below the waterfall will be lost as spawning habitat as it will be inundated by the reservoir, and the section above the waterfall will remain naturally inaccessible. It is likely that there is suitable habitat elsewhere within the upper catchment. However, further investigation will be required to determine the extent of this.

Productivity and juvenile trout rearing habitat within the reservoir will depend on the extent to which macrophyte beds establish in the littoral zone around the lake margins. Given the relatively low nutrient concentrations observed in water samples from the Lee River at the dam site the reservoir is likely to ultimately be relatively oligotrophic (see Section 4 above), although nutrient levels and consequent pelagic phytoplankton production may be higher initially, due to nutrients released from inundated vegetation and soils. Littoral plants and algae generally dominate primary production in oligotrophic water bodies (Vadeboncoeur *et al.* 2003), and the macroinvertebrate communities associated with littoral macrophyte beds are generally recognised to be the most diverse and productive component of the lake fauna (Kelly & McDowall 2004). Macrophyte beds can provide both cover and food for juvenile brown trout rearing in lakes.

The reservoir is likely to have a limited shallow littoral zone, due to the steep-sided nature of the valley. This, as well as fluctuations in water level, is likely to limit the development of macrophyte beds to some extent. Those plants that do establish around the shallow margins of the reservoir will periodically be exposed and will probably die off during periods of draw down. However, the incidence of the water level being drawn down to extremely low levels is expected to be relatively rare. The median annual maximum draw down that would have occurred during the synthetic record from January 1958 to November 2007 (*i.e.* the median over this period of the maximum reduction in water level behind the dam in each year) is approximately 2.5 m, while the 80^{th} percentile is approximately 6.5 m (*i.e.* the lake level would be expected to be drawn down by more than 6.5 m in only 20% of years). Overall the reservoir is expected to be full and spilling approximately 82% of the time and the water level is expected to be drawn down by more than 1 m only about 10% of the time. Given that the aim of the reservoir is to store and release water, water level variations are inevitable.

Consequently, there is likely to be scope for macrophytes to establish below the minimum water level maintained through most years, but within the reach of adequate light to support growth when the lake is full.

Schwarz *et al.* (2000) developed a regression model predicting maximum depth limits for rooted aquatic plants (Z_c) based on lake optical properties (diffuse attenuation coefficient, k_d)

and from 28 South Island lakes (including lakes in the Nelson region). This model takes the form:

$$Z_c = -2.5 + 4.7 / k_d$$

Assuming the newly formed reservoir will have similar inherent optical properties to Lakes Rotoiti and Rotoroa (k_d 0.26-0.28), with their similar beech forest dominated catchments, this model predicts the maximum depth limit for rooted plants to be 14.3-15.6 m. As stated above, the reservoir is expected to be drawn down by more than 2.5 m in only about 50% of years. Consequently, most of this depth zone with sufficient light to support plant growth is likely to be productive most of the time. If the upper 2.5 m of the littoral zone is discounted (due to its higher frequency of drying), a zone suitable for aquatic macrophyte growth can be expected between approximately 180.4 m and 193.5 m RL. Based on the predicted reservoir surface data available, this area of productive littoral zone is likely to be approximately 32% of the total surface area of the reservoir at maximum capacity (*i.e.* the surface area at 196 m RL) (Figure 13).

In addition, some macroinvertebrates in the littoral zone may be able to survive relatively short periods of exposure due to water level fluctuation and therefore the area maintaining food production for fish foraging in the littoral zone may be higher than implied by the above analysis. Logs, roots and moist vegetation on the exposed bed may provide some protection against desiccation (Winterbourn 1987). According to Greig (1973) some macroinvertebrates, such as snails and worms, inhabiting the upper littoral regions of Lake Waitaki were able to tolerate severe exposure, at least over the short-term. Furthermore, Fillion (1967) observed chironomid larvae surviving up to 85 days on an exposed lake shore. However, Greig (1973) noted that some animals, particularly caddis flies, did not appear in the littoral zone except at the bottom of the drawdown zone where water cover was more permanent and the finest sediment and organic matter was deposited.

Furthermore, during periods when the reservoir water level is drawn down to a minor extent it is possible that rapid colonising terrestrial margin species may flourish in the varial zone, and when this area is subsequently rewetted these plant may provide an energetic subsidy to aquatic food webs.

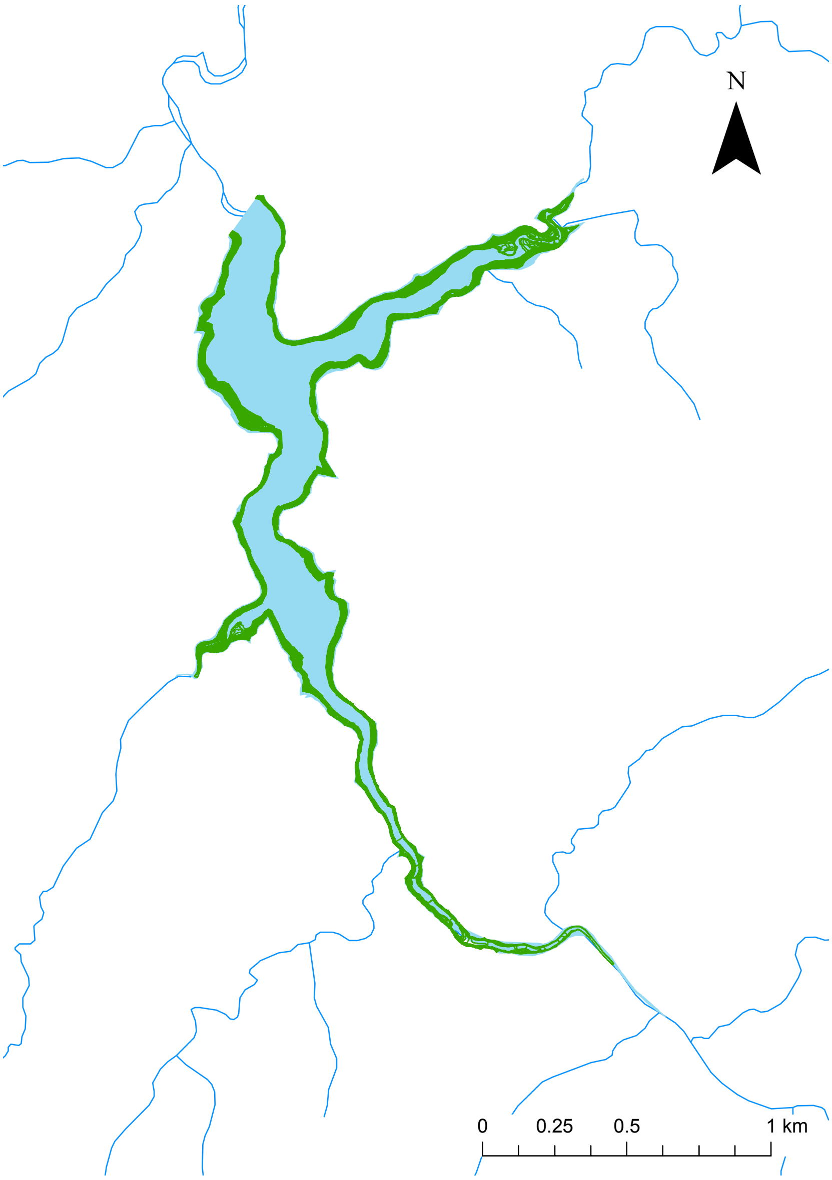

Figure 13. Proposed Lee Reservoir showing predicted extent of zone with sufficient light penetration to support macrophyte growth (shaded green).

Considering that about 32% of the lake surface area will be able to support aquatic plants, the storage lake may be able to support a relatively productive fishery, assuming that sufficient spawning habitat exists in the upper catchment to provide adequate recruitment. Similar information from other reservoirs is not easy to gather.

8.2. Nuisance macrophyte and periphyton growth

Excessive macrophyte growth has caused problems in regulated lakes in New Zealand, mainly attributable to non-native macrophyte species (Clayton & Champion 2006). Drifting masses of macrophytes can clog intakes and dense macrophyte growths can impair recreational values of reservoirs. Clayton & Champion (2006) devised a qualitative scoring system to assess the latent potential for aquatic weed impact in hydro-lakes. This system allocates a score for five physical characteristics of the water body, which are related to the potential for development of nuisance aquatic weed growths. Clayton & Champion suggested that any lake scoring above 15 has the potential to support significant growths of submerged vegetation and could present some inconvenience to power production (*e.g.* through clogging intakes).

Table 10 shows an application of this scoring system to the proposed Lee reservoir. For this application we have used the median annual maximum draw down of 2.5 m, as indicative of the level fluctuation to be expected from year to year. Obviously fluctuation will be greater than this in some years, with resulting reductions in macrophyte biomass. But by the same token, in other years there will be little or no water level fluctuation.

The water clarity recorded from the Lee River in the vicinity of the dam is in the order of 7–10 m. On the assumption that water clarity in the lake will be similar we have adopted a score for water clarity of 5.

The proposed reservoir is relatively open at the downstream end, although it becomes more channelised at the upper end but with some meandering. However, the prevailing wind is likely to be up-valley in the summer, either through channelisation of west to northwest winds, or through anabatic thermally developed winds in summertime. Both the meandering nature of the reservoir and the likely prevailing wind direction may act against dislodged weed masses moving directly toward the outlet, where they would likely cause clogging. On this basis we have adopted an intermediate score of 3 for shoreline shape.

The Lee Valley in the vicinity of the proposed dam is relatively steep sided. Hence we have applied a score of 1 for the littoral gradient and substrate risk factor. However, the reservoir is relatively small, and consequently the fetch for wave development is relatively small in any direction, and the meandering shape of the reservoir exacerbates this. Clayton & Champion suggest that low wave fetch makes macrophyte beds more susceptible to mass uprooting events during occasional storms. We have adopted a score of 3 for this factor. However, it could be argued that a score of 4 may be appropriate.

This assessment suggests that there is likely to be some risk of macrophytes causing clogging issues for the outlet (Table 10), although the risk is toward the low end of the scale (*i.e.* 15 or 16, just over the suggested threshold for concern). A mitigation plan may be required to address this potential issue.

Another issue that has been raised is the potential impact of the invasive alga, didymo (*Didymosphenia geminata*). This alga does not generally proliferate in lakes, although masses of senescent cells may be flushed into lakes from growths in the tributary rivers or stream, and may then drift into intakes and cause clogging issues. The densely bush-clad nature of the upper Lee catchment means that algal proliferation in the upper catchment is unlikely. However, any management plan aimed at reducing clogging of the outlet with macrophyte masses is likely to also address potential clogging issues associated with didymo, albeit that these are unlikely to occur.

Table 10. Risk assessment scoring system for submerged weeds in New Zealand hydroelectric lakes from Clayton and Champion (2006), and scores selected for proposed Lee Reservoir.

Physical characteristics		Scores		Selected Score
A	Level fluctuation	1	Fluctuation >6 m	
		2	Fluctuation 4-6 m	
		3	Fluctuation 2-4 m	3
		4	Fluctuation 1-2 m	
		5	Fluctuation <1 m	
B	Water clarity	1	Turbid – Secchi <0.5 m	
		2	Secchi 0.5-2 m	
		3	Secchi 2-3.5 m	
		4	Secchi 3.5-5 m	
		5	Clear water - Secchi >5 m	5
C	Shoreline shape	1	Open lake	
		3	Relatively open lake with outlet receiving prevailing winds	3
		5	Channelised	
D	Littoral gradient and substrate	1	Steep, rocky gradients	1
		3	Moderate gradients with sand/silt sediments	
		5	Shallow shelving shoreline with silty sediments	
E	Exposure to wave action	1	Prevailing wave fetch >4 km over 50% of shoreline	
		3	Prevailing wave fetch <4 km	3
		5	Sheltered small lake <1 km max dimension, low wind/wave impact	
			Total	15

9. ACKNOWLEDGEMENTS

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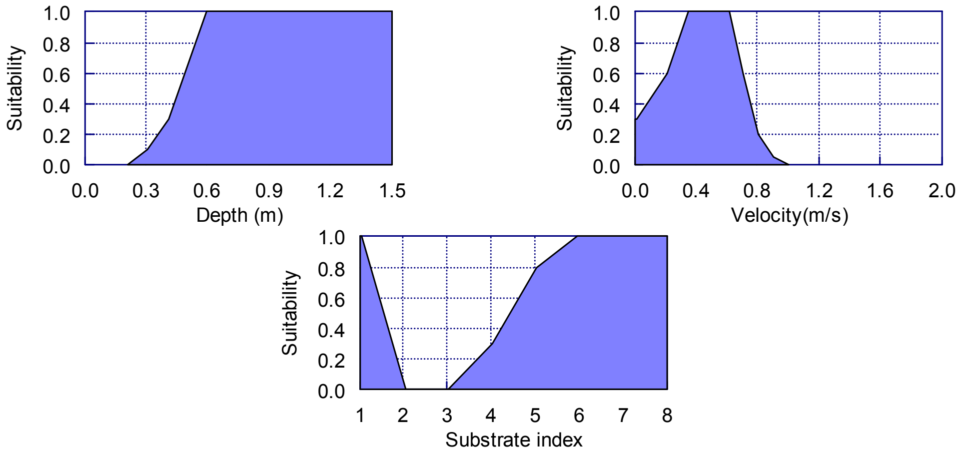
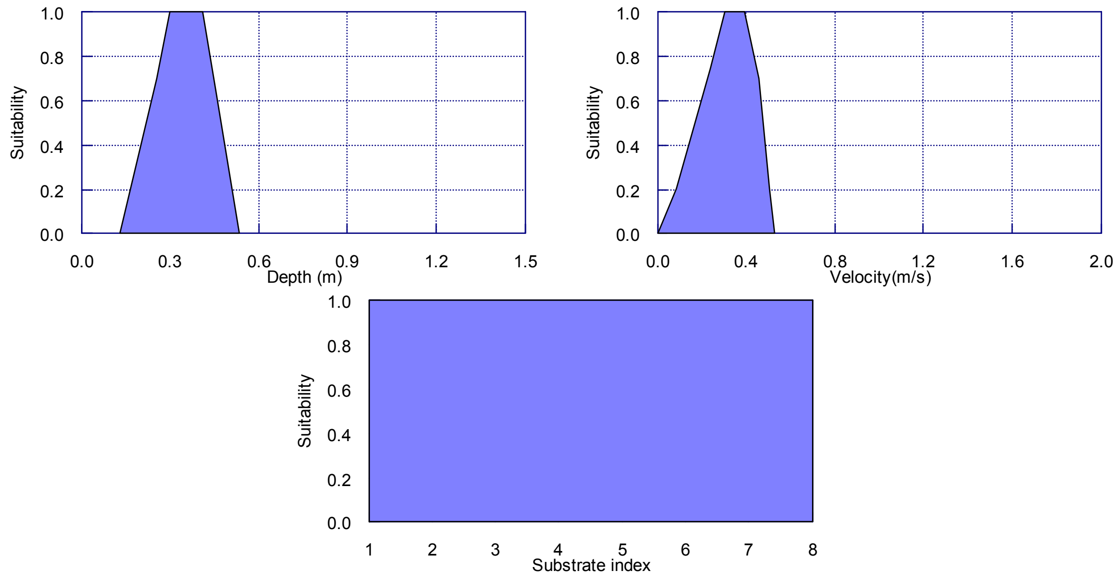
11. APPENDICES

Appendix 1. Habitat suitability criteria used in this report

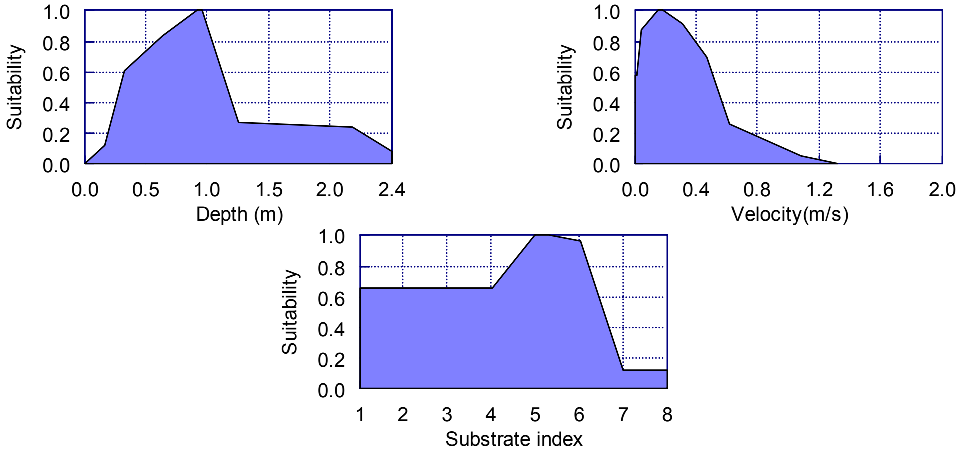
Brown trout adult (Hayes & Jowett 1994)



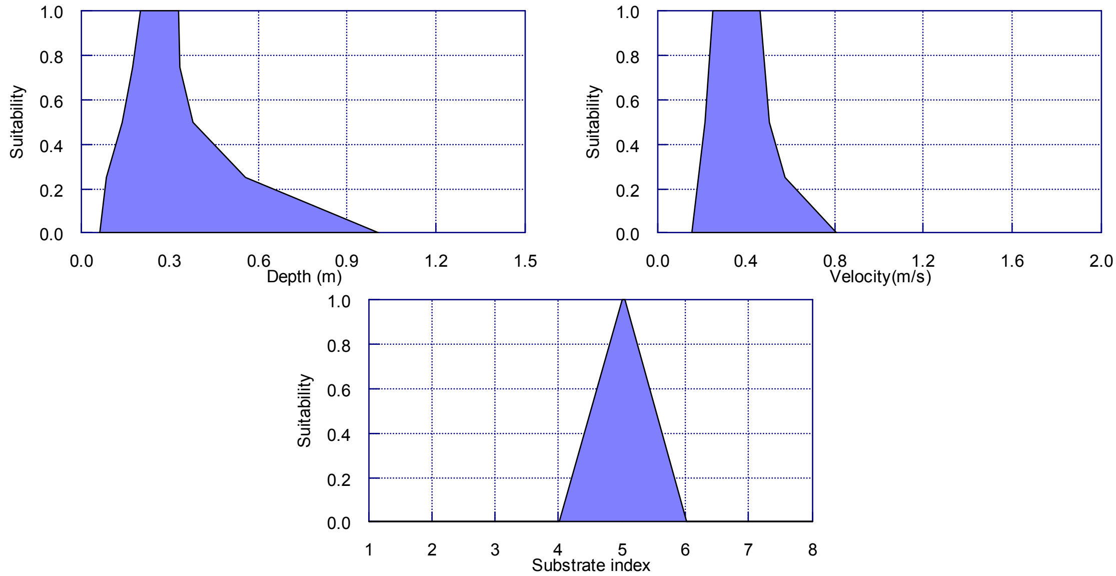
Brown trout yearling - small adult feeding (Roussel et al 1999)



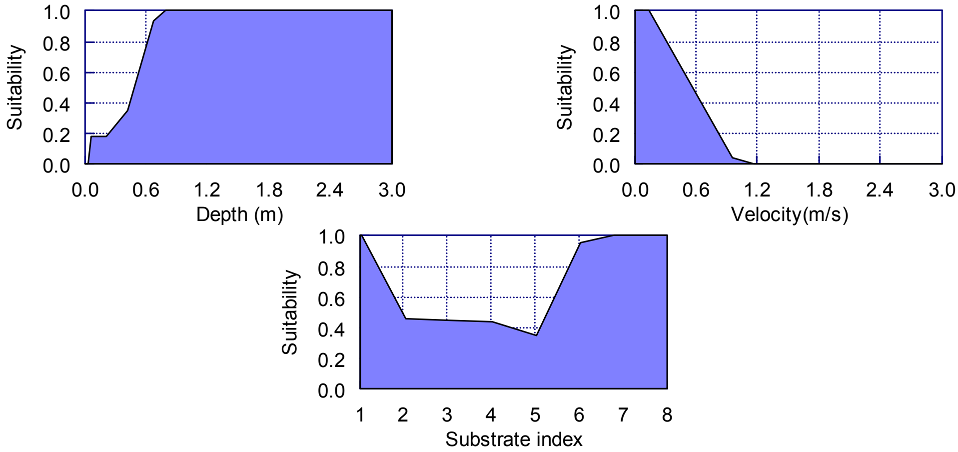
Brown trout 15-25cm (Raleigh et al. 1986)



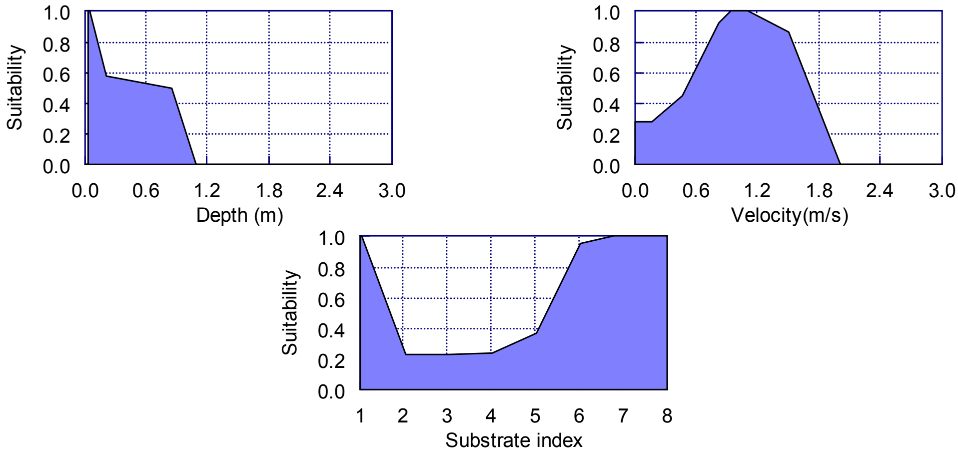
Brown trout spawning (Shirvell & Dungey 1983)



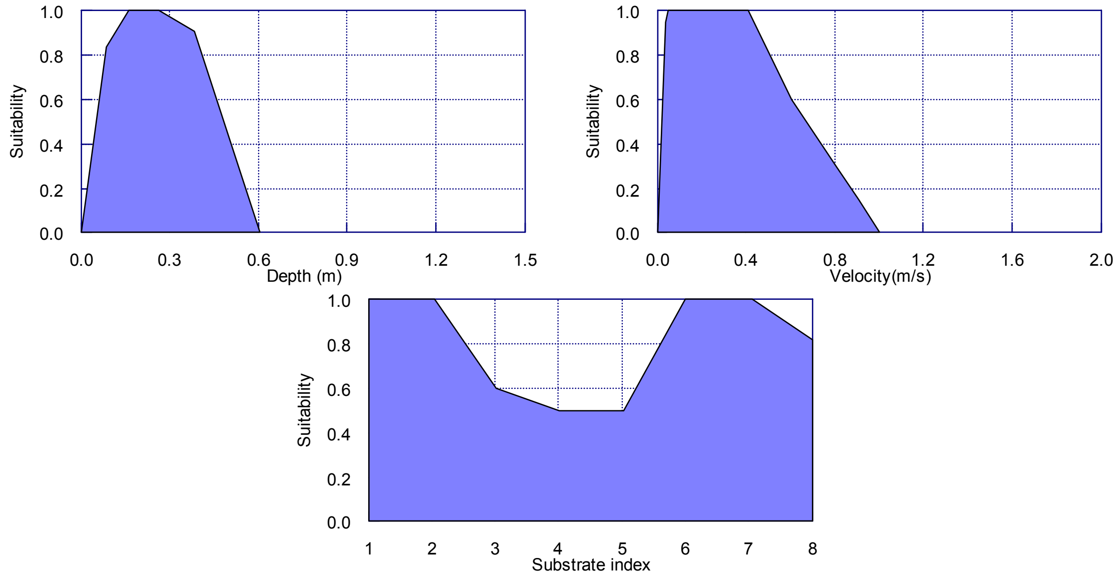
Longfin Eel > 300 mm (Jellyman et al. 2003)



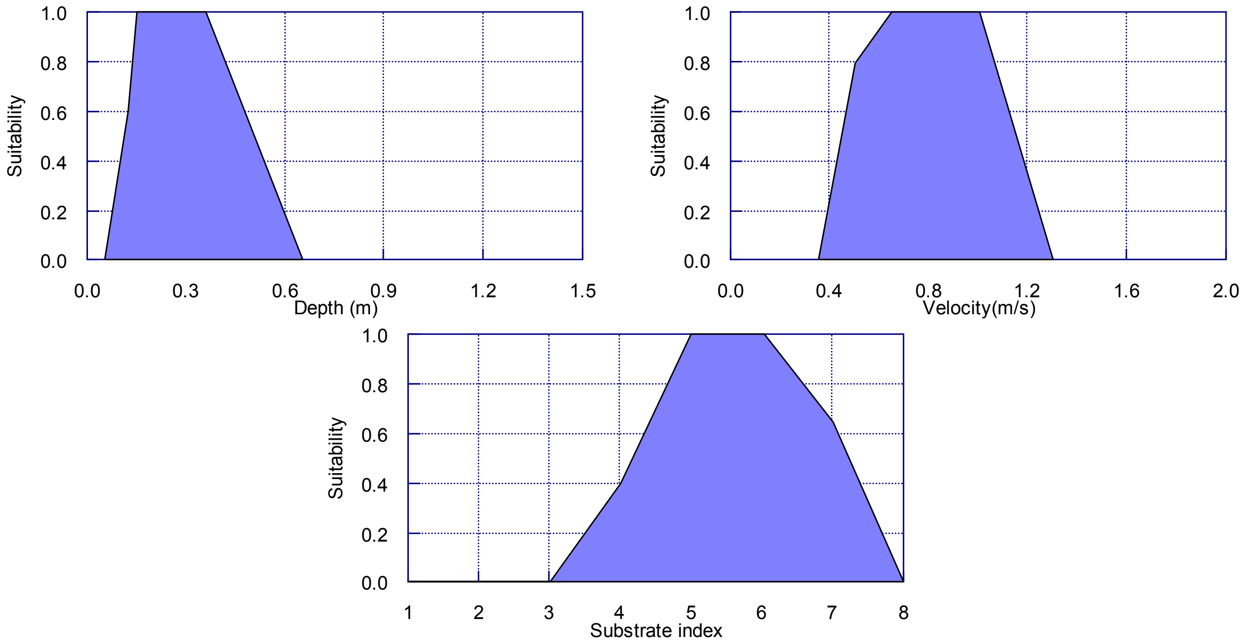
Longfin Eel < 300 mm (Jellyman et al. 2003)



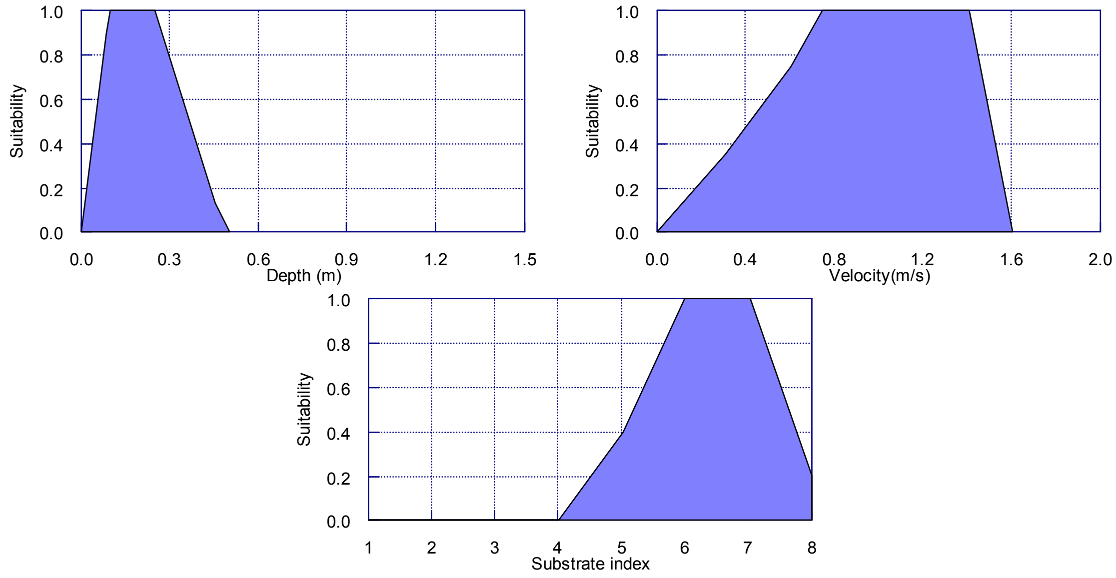
Shortfin eel < 300mm (Jowett & Richardson 2008)



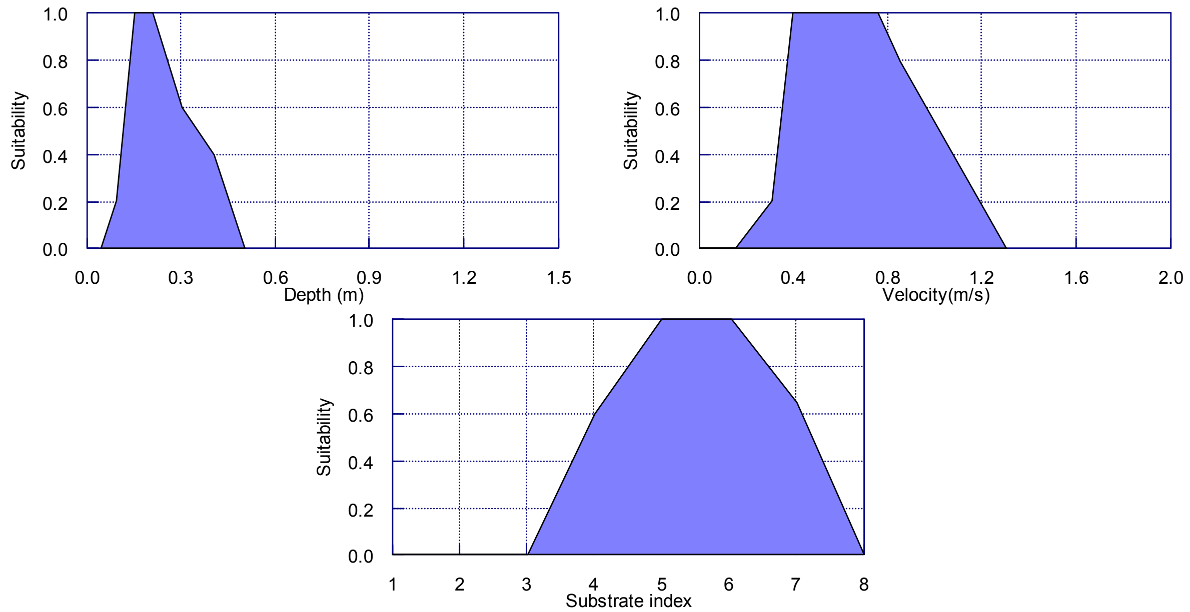
Torrentfish (Jowett & Richardson 2008)



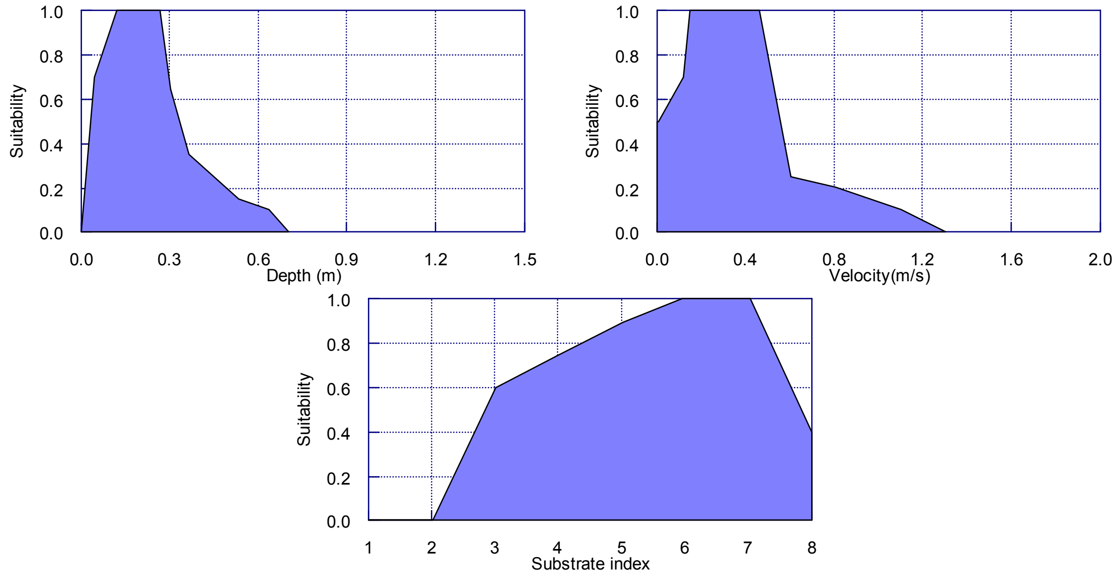
Koaro (Jowett & Richardson 2008)



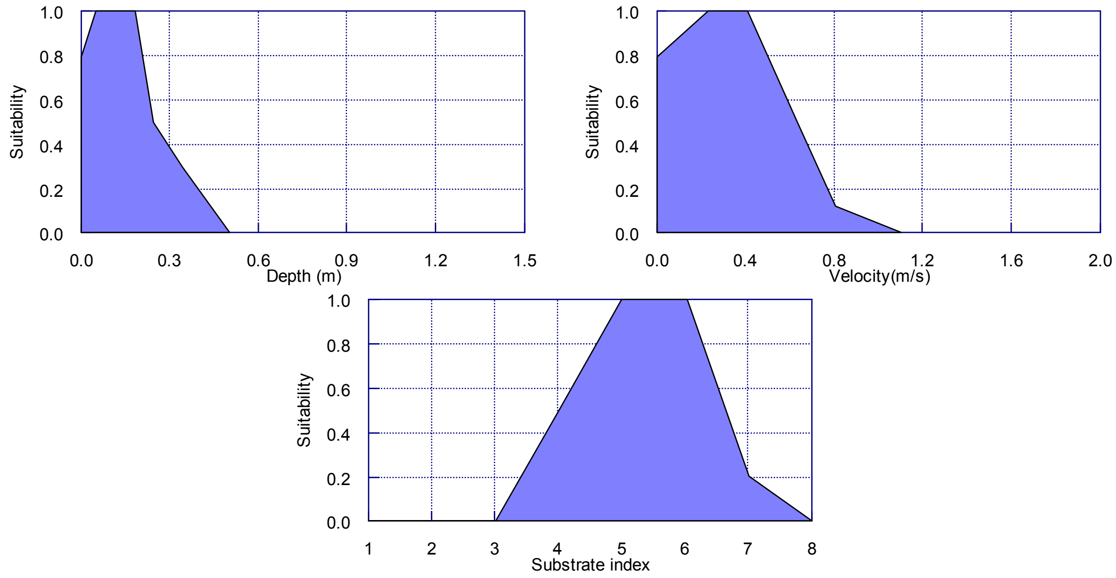
Bluegill bully (Jowett & Richardson 2008)



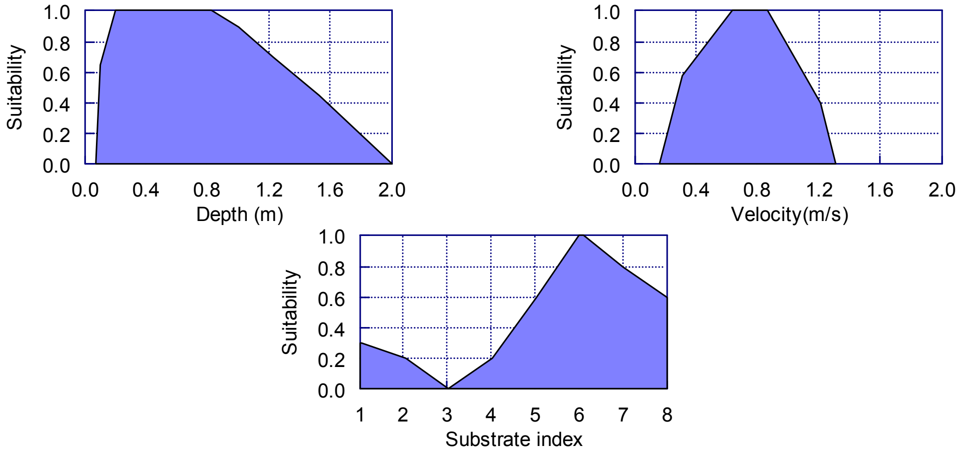
Redfin bully (Jowett & Richardson 2008)



Upland bully (Jowett & Richardson 2008)



Food producing (Waters 1976)



Appendix 2. Variation in predicted habitat quality (HSI) for a) trout and b) native fish in the Lee River (Upper and Meads Bridge reaches combined). Blue dashed line denotes 7-day MALF.