

September 2008 Juvenile Rainbow Trout Abundance in the Sutherland River System, British Columbia

John Hagen¹



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Smithers, British Columbia

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¹ J. Hagen and Associates: 330 Alward Street, Prince George BC, V2M 2E3, hagen_john2@yahoo.ca

ABSTRACT

Lacustrine populations of large-bodied, piscivorous rainbow trout in British Columbia are relatively small, and highly vulnerable to exploitation pressure. Babine Lake sport fisheries and a First Nations gillnet fishery have for many years targeted a population of large rainbow trout whose natal stream is the Sutherland River at the lake's southeastern end. Because the current status of this population is unknown, the British Columbia Ministry of Environment in 2008 initiated a population monitoring study. Monitoring of spawner abundance is not currently feasible, so juvenile rearing habitats of the Sutherland River system were sampled instead using electrofishing in enclosed sites. To increase sampling efficiency, a single-pass electrofishing methodology was employed, with first-pass catches calibrated against three-pass removal abundance estimates at a portion of the total number of sites. Electrofishing sites included those approximate locations sampled during a baseline inventory study in 1988, to increase the power of comparisons across the twenty-year time gap. Juvenile rainbow trout dominated electrofishing catches at all locations. Based on the age structure of the sampled population, the dominant age of lake entry appears to be age-3, with a small number of precocious male parr remaining in the system for a fourth year. First-pass capture efficiency, relative to three-pass abundance estimates at calibration sites, was high and relatively precise across the most commonly sampled age classes 0+ to 2+. Juvenile rainbow trout production, from the portion of the system that could be electrofished effectively, was generally comparable to 1988 estimates (22,901 versus 20,484 for age-2+ parr in September 2008 versus September 1988, respectively), suggesting either comparable levels of brood spawner abundance or density-dependent population regulation resulting from adequate seeding of juvenile rearing habitats. These results are consistent with the notion that the population's status is not greatly degraded relative to 1988, but increasing confidence in this conclusion will require additional years' sampling data. Comparisons of empirical density estimates with modeled predictions of maximum density were less clear. Good agreement was exhibited between sampling data and predictions from a general salmonid model of maximum density based on fish size and total alkalinity (within 4%), but estimated maximum density was only 61% of predictions from a more recent model developed for coastal cutthroat trout streams.

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1.0 INTRODUCTION

The importance of maintaining genetic and ecological diversity below the species level has been widely accepted in conservation management (Leary et al. 1993; Moritz 1994; Waples 1995). Phenotypic and genetic diversity of rainbow trout (*Oncorhynchus mykiss*) in British Columbia appear to be consistently associated with the habitats that populations inhabit (Keeley et al. 2005). Among ecotypes studied by Keeley et al. (2005), lake-dwelling populations of large-bodied, piscivorous rainbow trout appear to be the most rare. Fewer than 20 such populations occur in British Columbia, generally in lakes of surface area >2,000 hectares which also support kokanee (De Gisi 2003). In addition to their morphological, ecological, and genetic uniqueness, lacustrine populations of large, piscivorous rainbow trout are of high recreational, economic, and cultural value, and are therefore a top management priority for BC's Ministry of Environment. Because of high exploitation pressure and relatively small populations, easily accessible lakes inhabited by piscivorous rainbow trout populations have had harvest opportunities restricted in recent years.

A lone population of large, piscivorous rainbow trout has been described to date from Babine Lake in northcentral BC. These fish, which can reach 8 kg in size (Bustard 1990), appear to preferentially utilize the southern end of Babine Lake, and spawn in the Sutherland River. Following habitat use studies in the late 1980s (Bustard 1989, 1990), angler harvest regulation changes protecting the population were introduced by the BC Ministry of Environment. Currently, regulations permit harvest of only one fish >50 cm per day, and 12 km of Babine Lake extending from the mouth of the Sutherland River are closed all year, (BC freshwater angling regulations synopsis, 2009-2010).

In addition to Babine Lake sport fisheries, Sutherland River-origin rainbow trout have been targeted in a Yekooche First Nation gillnet fishery for pre-spawning adults near the mouth of the Sutherland River, which for many years has been conducted for food, social and ceremonial purposes. Recent anecdotal reports have identified that the Yekooche netting efforts (and harvest) have been highly variable from year to year (Mark West, Burns Lake Conservation Officer Service, pers comm.). During a radio telemetry study of spawner habitat use in spring, 1989, in which tangle nets were employed to capture fish prior to tagging, the gillnet fishery was also monitored (Bustard 1990). A mark-recapture population estimate, based on fish recaptured during tangle netting, suggested that the run comprised roughly 500 spawners. At least 49 rainbow trout spawners were harvested in the Yekooche gillnet fishery, suggesting an annual exploitation rate of 10% or more from that fishery alone. This was thought to be a conservative estimate of annual exploitation at that time, as native fishers indicated that fishing effort was lower than normal in 1988 (Ibid.). Although not confirmed, recent gillnetting effort and harvest are thought to be reduced from historical levels due to unknown factors (Mark West, Burns Lake Conservation Officer Service, pers comm.).

Although native and recreational fisheries continue, the current status of Babine Lake's piscivorous rainbow trout population is unknown. The conservation status of a fish population, or group of related populations, is usually evaluated as the likelihood of

its long-term persistence. Population data usually evaluated in assessing status are total adult abundance relative to recommended thresholds, the trend of abundance over time, population spatial structure, genetic diversity, and the existence of habitat conditions that threaten populations (McElhany et al. 2000).

An alternative approach for assessing population status has been suggested by BC's Ministry of Environment for closely-related steelhead (anadromous *O. mykiss*) populations, whereby biologically-based reference points of stock size provide a framework for assessing the need for management changes (Johnston et al. 2000). Ideally, reference points are based on a stock-recruitment relationship for the watershed.

Estimating spawning stock for the Sutherland River rainbow trout population is not currently feasible. Neither a time series of spawner population estimates, therefore, nor even a single year's snapshot can be reasonably estimated for comparison with guidelines emerging from the conservation biology literature (McElhany et al. 2000), or with juvenile abundance estimates to generate an empirical stock-recruitment relationship. Reliable estimates of juvenile rainbow trout abundance, however, derived from electrofishing in enclosed sites, have proven to be feasible at most locations in the Sutherland River other than beaver dam impoundments or the lowest reach of the river, which is deep and silty (Bustard 1989).

For populations dependent on a period of tributary rearing, during which juvenile production is likely to be density dependent, a natural management reference point that can be derived from juvenile abundance data alone is the abundance of juvenile trout that fully seeds available rearing habitats. This will be indicated by the asymptote that juvenile abundance estimates approach in time series data. Populations will, in theory, exhibit increased stability if juvenile abundance is close to this asymptote – neither higher nor lower spawner stock sizes will increase juvenile recruitment unless egg deposition drops below that necessary to seed the available rearing habitats.

Funding for this project was uncertain beyond a single year of sampling, meaning time series data beyond 2008 and 1988 (the only other year in which sampling for juvenile rainbow trout abundance has taken place; Bustard 1989) would potentially not be available. A large database of juvenile abundance estimates have been used by the BC Ministry of Environment to develop empirical relationships between maximum salmonid densities in typical sampling sites (roughly 100 m²), fish size, and water quality parameters (Ptolemy 1993; Ptolemy 2005). In theory, predictions of maximum density provide a benchmark for comparison with observed maximum densities from a single sampling period, as an alternative to evaluating time series data.

This report documents the results of a juvenile rainbow trout abundance study in the Sutherland system conducted in September 2008 by the BC Ministry of Environment. The primary objective of the study was to investigate potential changes in the population's status, by collecting juvenile trout abundance data utilizing electrofishing in enclosed sites for comparison with 1) 1988 rainbow trout population density estimates (Bustard 1989), and 2) with predictions of maximum density (Ptolemy 1993).

2.0 METHODS

2.1 Study Area

The Sutherland River watershed is described in Bustard (1989), which provides reach lengths and watershed areas estimated from 1:50,000 NTS mapping. The Sutherland enters Babine Lake at its southern end (Figure 1), and is the third largest tributary to Babine Lake, after the Fulton River and Pinkut Creek. The majority of the Sutherland Watershed is contained within either a Class A Park (in the lower watershed) or a Protected Area (in the upper watershed). The Sutherland has a mainstem length of approximately 70 km between Babine and Sutherland Lakes, which is of relatively low gradient. Stream gradient downstream of Duncan Creek (Figure 1) is less than approximately 1%, but steepens to over 2% in upper sections above Duncan Creek (Bustard 1989). It is uncertain whether numerous beaver dams in the upper section of the Sutherland River restrict access for migrating rainbow trout spawners during spring. Incidental redd observations during this study confirmed spawner utilization at least as far upstream as site DBS7 37.0 (Figure 1), which was also identified by Bustard (1989) as a likely area of localized spawning based on high fry abundance.

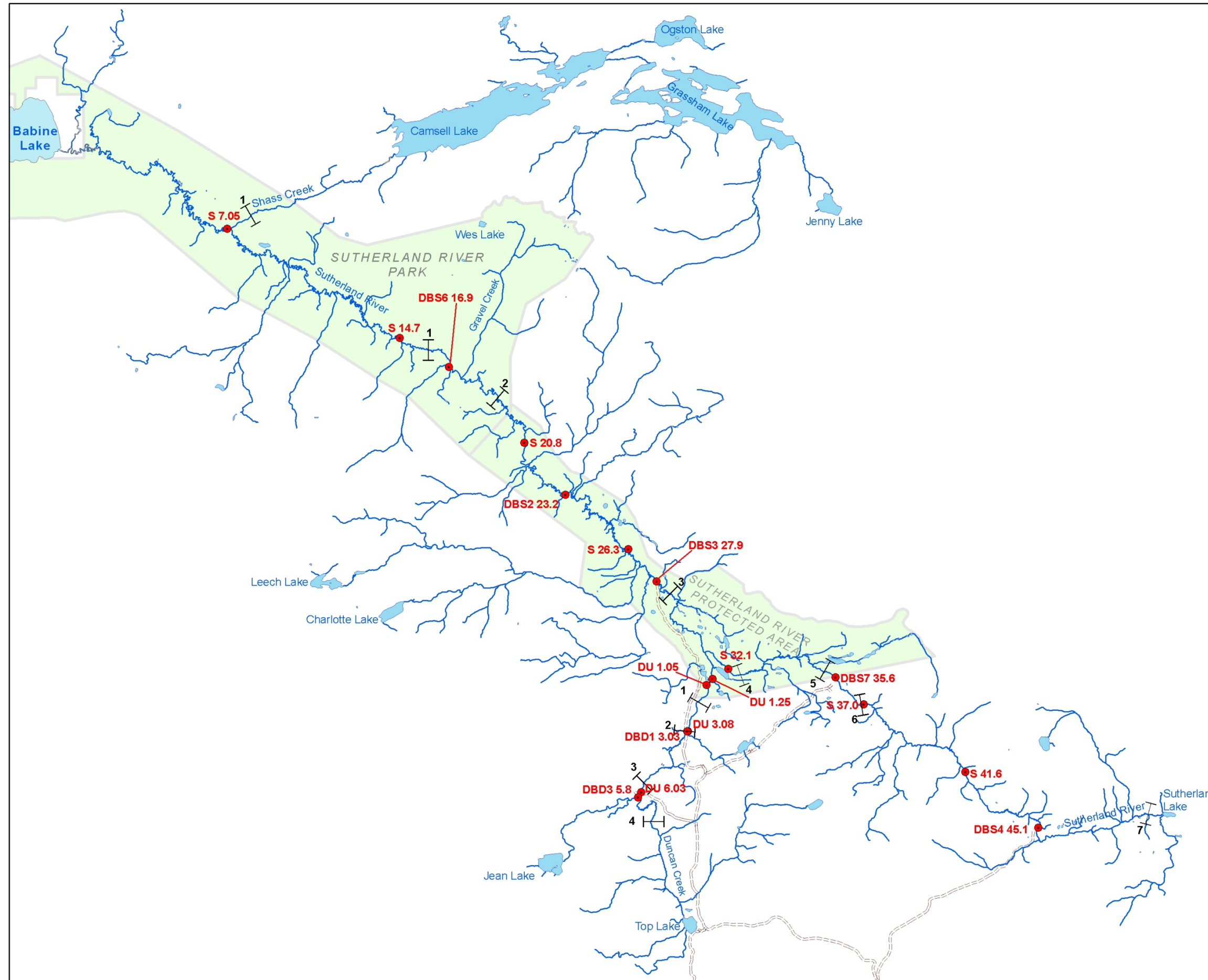
Four of the seventeen (24%) radio-tagged rainbow trout adults that entered the Sutherland River during the 1989 habitat use study (Bustard 1990) appeared to utilize Duncan Creek for spawning, indicating the potential importance of this tributary for spawning and rearing. Potential habitat use by adfluvial rainbow trout in Duncan Creek was estimated to be approximately 7 km by Bustard (1989). Tributaries other than Duncan Creek are likely to be of little importance for adfluvial rainbow trout, given their small size, steep gradient, and limited extent of spawner access. Shass Creek and Gravel Creek (Figure 1) have potential for spawning and rearing rainbow trout, but access is limited to 1 and 1.5 km, respectively. Other species reported for the Sutherland system have been mountain whitefish (*Prosopium williamsoni*), kokanee and sockeye salmon (*Oncorhynchus nerka*), coho salmon (*Oncorhynchus kisutch*), prickly sculpins (*Cottus asper*), and a lone brassy minnow (*Hybognathus hankinsoni*; Bustard 1989) which may have been a misidentified lake chub (McPhail 2007).

Water Survey of Canada flow data are not available for the remote Sutherland system. Based on an estimated watershed area of 656 km², and water yield/km² estimates for other systems in the region thought to be of comparable hydrology (Pinkut Creek, Maxan Creek, and Buck Creek), predicted mean annual discharge and September discharge at the mouth are 3.9 m³/s and 1.95 m³/s, respectively (Ron Ptolemy, BC Ministry of Environment, Victoria, data on file).

2.2 Study design

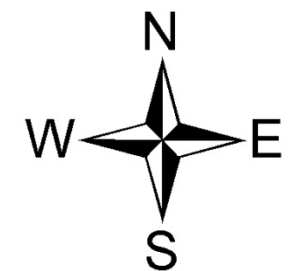
The error associated with spatial variation in fish density within a stream (first stage error; Hankin 1984) is generally much greater than the measurement error in smaller streams associated with estimating fish abundance at each site (second stage error). Stream fish populations are often highly clumped in their distribution as a result of spawning distribution (Beard and Carline 1991), geomorphic influences such as

Figure 1 (following page). Sites sampled in the Sutherland River system in 2008. Depicted reach breaks are according to Bustard (1989); the mainstem Sutherland River channel could not be spanned with stop nets in reach 1.



Sutherland Watershed Juvenile Rainbow Trout Survey Sites

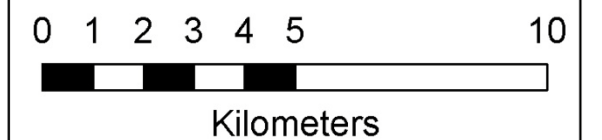
September 2008



Legend

- Sample sites
- Streams
- Reach breaks
- Lakes
- Roads
- Protected Areas

Projection : BC Albers
Scale 1:150,000



elevation, channel slope, and stream size (Kruse et al. 1997), habitat variability (Newman and Waters 1984), intercohort competition (Bohlin 1978), channel alterations (Elser 1968), or other factors. Research in smaller streams has suggested that, as a result of high spatial variation in fish abundance, sampling a greater number of sites using a quicker but less precise method can provide greater precision in an abundance estimate overall for a given cost (Hankin and Reeves 1988; Decker et al. 1999). Hankin and Reeves (1988) recommended a strategy whereby a relatively quick method of population assessment, which is applied at a large number of sites, is calibrated results from a more accurate but time-consuming method at a portion of the sites.

Population estimates derived from multiple-pass, removal electrofishing have been the standard for quantitative juvenile salmonid studies in small streams (Ptolemy 1993; Riley et al. 1993; Peterson et al. 2004; Rosenberger and Dunham 2005), and, for the Sutherland system, form the basis for comparison with the results of this study (Bustard 1989). First-pass electrofishing catches, however, are typically good predictors of population estimates derived from multiple passes (Lobon-Cervia and Utrilla 1993; Crozier and Kennedy 1994; Jones and Stockwell 1995). In this study, we utilized single-pass electrofishing in enclosed sites as a ‘rapid assessment’ method (Jones and Stockwell 1995), calibrated by three-pass removal estimates at a portion of the sites. We considered the improved sampling efficiency of this approach, over the use of multiple-pass electrofishing alone, to be particularly important given the relatively high cost and necessity of helicopter travel in this remote watershed.

Reconnaissance of the Sutherland system prior to the beginning of fieldwork suggested that stratification of sampling based on habitat type classification, such as ‘riffle’ or ‘pool,’ was unnecessary given the habitat complexity and relatively short length of habitat units. We systematically sampled three strata that we expected would explain meaningful variation in fish density within the Sutherland system, based on prior sampling data (Bustard 1989): 1) the lower 25 km of the Sutherland River (Figure 1: downstream of DBS6 16.9; reach 1 in Bustard 1989), which is slow and silty, relatively difficult to sample using electrofishing, and likely to be of relatively lower importance for rainbow trout fry and parr; 2) the remaining 45 km of the Sutherland River mainstem below Sutherland Lake; and 3) the accessible, lower 7 km of Duncan Creek, which was identified in Bustard (1989) as being of particular importance to spawning and rearing rainbow trout. The relatively simple stratification scheme was designed to allow adequate replication in strata 2 and 3 above for estimation of juvenile rainbow trout standing stock. Reaches were sampled systematically by spacing sampling sites roughly equal distances apart (pairs of sampling sites in the case of Duncan Creek, which had fewer access options). Locations sampled previously (Bustard 1989) were incorporated into the sampling design to facilitate comparison, although these could not be located precisely in most cases because position coordinates were not available.

2.3 Fish sampling

Electrofishing sites were fully enclosed (one bank to the other) by upstream and downstream stop nets suspended by aluminum bi-pods (Figure 2). The exception to this

was lower Sutherland reach (Figure 1), where sampled sites were too wide and deep to be enclosed. The upstream of two sites sampled in this reach (Site S 14.7; Figure 1) was not suitable for stop nets and was sampled as an open site. Only a single bank of the lower site (S 7.05) was sampled, and only a bottom stop net could be deployed. At S 7.05 the offshore side of the site was bounded by water too swift to be utilized by age-0+ rainbow trout, and electrofishing proceeded always from the fast water forming the offshore boundary towards the shore, to avoid chasing juvenile trout from the site.

At all sites, an electrofishing pass was initiated at the downstream net, and consisted of a thorough search in an upstream direction, followed by a systematic sweep back towards the downstream net. If three electrofishing passes were to be conducted, at calibration sites, the site was rested for approximately one hour between passes, during which fish captured during the preceding pass were sampled and site habitat data collected. As much as possible, calibration sites were electrofished in an identical manner on each pass (minus fish handling time), as required by the assumption of constant capture probability among passes (Zippin 1956).

All salmonids captured during single-pass and three-pass electrofishing were anaesthetized, identified as to species, measured for fork length (nearest mm), held in pens and then released back into the site following the completion of sampling. At a portion of the total number of electrofishing sites, all fish captured from the site were weighed (nearest 0.1 g) to enable development of a length-weight relationship. Scale samples were also collected from a portion of the juvenile rainbow trout captured to assist with aging analysis. Fish ages were determined by Carol Lidstone of Birkenhead Scale Analyses (Lone Butte, BC) by inspecting scales mounted on glass slides. The best scale for each individual fish is photographed with a digital microfilm scanner and the age determination is completed by counting the annuli. Annuli are zones of closely spaced, thin and/or incomplete circuli which indicate a sudden decrease in growth rate and represent the period of winter growth (Carol Lidstone pers. com.).

Tissue samples were collected from rainbow trout fry and parr at a number of electrofishing sites for later genetic analyses (nDNA), as a contribution to a concurrent study of population structure among tributaries to Babine Lake (Koehler 2010). A small number of age1+ and older rainbow trout parr were sacrificed to assess their maturity status (Figure 3), and assess alternative life history strategies within the population(s).

Physical site attributes recorded at sampling sites included descriptions of substrate composition (boulder, cobble, gravel, and fines as percentages of the site area), D90, D50 (diameters of substrate particles for which 90% and 50%, respectively, of the site area consist of smaller particles), site length, wetted width, channel width, percent cover (categories included: overhead vegetation, turbulence, deep water and boulder as percentages of the site area; undercut bank as a percentage of the combined length of the stream banks, and the total area of wood debris > 10 cm in diameter), and average and maximum thalweg depths (based on 10 measurements, each taken at the deepest point along a cross-channel transect). Measurements of stream conductivity were taken at a several sites in Duncan Creek and the Sutherland River, and water temperature measurements at every site.



Figure 2. Electrofishing site DBS2 23.2 located below Duncan Creek but above Reach 1 of Bustard (1989).



Figure 3. Mature male rainbow trout parr from the Sutherland River system, September 2008.

2.4 Analyses

2.4.1 Length-at-age and the length-weight relationship

Length-age cutoffs for age classes of juvenile rainbow trout in the Sutherland River system were established by inspecting length-frequency histograms, and corroborated with estimated ages from scale analysis. Length-frequency histograms included length data from all fish captured by electrofishing in the reach of interest. The only scale age data used in corroborating length-age cutoffs was that taken from scales that could be clearly interpreted. Length-age cutoffs were established separately for the upper Sutherland, Duncan, and lower Sutherland (mainstem below Duncan Creek) stream sections, to match observed patterns of variation in length-at-age across the Sutherland system.

The length-weight relationship for Sutherland system juvenile rainbow trout was estimated through simple linear regression of \log_{10} -transformed weight data on associated \log_{10} -transformed fork length measurements (Le Cren 1951). The resulting linear equation was then utilized to estimate the weight of all juvenile rainbow trout that had been measured for fork length but not weighted, which permitted estimation of total biomass for each age class across sites.

2.4.2 First-pass electrofishing capture efficiency

Previous estimates of juvenile rainbow trout abundance in the Sutherland River system have been based on multiple-pass electrofishing utilizing the removal method (Seber and Le Cren 1967; Bustard 1989). Multiple-pass electrofishing was also the most common source of abundance data for developing the models predicting maximum salmonid density in streams from fish size and water quality parameters (Ptolemy 1993; Ptolemy 2005). We therefore calibrated our rapid assessment method, single-pass electrofishing, with three-pass electrofishing at a portion of the total number of sampled sites in order to facilitate comparisons of abundance data with previous sampling results, and also with maximum density model predictions. Underestimation bias in the multiple-pass, electrofishing removal methodology has been well documented, however, as a result of violations of key assumptions in the approach (Bohlin and Sundstrom 1977; Peterson and Cederholm 1984; Riley et al. 1993; Peterson et al. 2004). Although useful for comparisons presented here, estimates of first-pass electrofishing efficiency are therefore relative and do not account for underestimation bias in the removal method of population estimation.

As the first step in the analysis of three-pass, removal electrofishing data from calibration sites, maximum likelihood (ML) estimates of abundance at each of the sites, standard errors of the ML estimates, and upper and lower 95% confidence limits were computed for each age class. Computations utilized formulae in Zippin (1956), which assume constant capture probability across passes at an individual site. Maximum likelihood estimates of capture probability across all passes, standard errors of the ML estimates, and upper and lower confidence limits for capture probability (Zippin 1956) were also computed for comparison among calibration sites and age classes.

For each age class (k), maximum likelihood estimates of first-pass capture efficiency $P(cap)_1$ at individual calibration sites (j) can be estimated as:

$$P(cap)_{1,j,k} = \frac{C_{1,j,k}}{N_{ML,j,k}} \quad (1)$$

where C_1 is the first-pass catch, and N_{ML} is the ML estimate of abundance based on three-pass electrofishing data. Estimates of $P(cap)_1$ are proportions, and therefore assumed to be binomially distributed. For each age class, therefore, the maximum likelihood estimate of mean first-pass capture efficiency $\hat{P}(cap)_1$ was estimated as the value that maximized the binomial probability (sum of the binomial log-likelihoods; Haddon 2001) of the observed first-pass catches across all calibration sites, given the three-pass ML estimates of abundance. $\hat{P}(cap)_{1,k}$ was estimated in Excel (Microsoft Corp.) using the Solver non-linear iterative search routine.

Limits of 95% confidence for the ML estimate of mean first pass capture efficiency were estimated using a deterministic approximation to the method of likelihood profile. Expected log-likelihoods for confidence limits for $\hat{P}(cap)_1$ are given in Haddon (2001) by:

$$2 \times [LL(\hat{P}(cap)_1)_{Max} - LL(\hat{P}(cap)_1)] \leq \chi^2_{1,1-\alpha} \quad (2)$$

$$LL(\hat{P}(cap)_1) = LL(\hat{P}(cap)_1)_{Max} - \frac{\chi^2_{1,1-\alpha}}{2} \quad (3)$$

where $\chi^2_{1,1-\alpha}$ is the $(1-\alpha)^{th}$ quantile of the χ^2 distribution with 1 degree of freedom (i.e. for 95% CL's $\alpha = 0.95$, $1-\alpha = 0.05$, and $\chi^2_{1,1-\alpha} = 3.84$). To estimate 95% confidence limits for mean first-pass capture efficiency, therefore, the Solver routine in Excel was utilized to search for the values of $\hat{P}(cap)_{1,k}$ that generated log-likelihoods equal to the maximum likelihood estimate minus half the required χ^2 value ($= 1.92$; equation 3). Estimates of standard error for each of the ML estimates of mean first-pass capture efficiency were approximated as the average half confidence interval divided by 1.96, noting that by the Central Limit Theorem the distribution of means from a non-normal population will tend towards normality as sample size increases (Zar 1996).

2.4.3 Juvenile rainbow trout abundance estimates

Maximum likelihood estimates of juvenile rainbow trout abundance (N) for each age class (k) at individual sampling sites (j) were calculated as:

$$N_{j,k} = \frac{C_{1,j,k}}{\hat{P}(cap)_{1,k}} \quad (4)$$

where C_1 is the first-pass catch at the site, and $\hat{P}(cap)_{1,k}$ is the ML estimate of mean first pass capture efficiency for the age class. As described above, 95% confidence limits for abundance estimates at each site were estimated using the Solver routine in Excel, by searching for the values of $N_{j,k}$ that generated a sum of the log-likelihoods equal to the maximum likelihood minus half the required χ^2 value (equation 3), given the first-pass catch $C_{1,j,k}$ and the calibration data for that age class. Site density estimates were calculated both on a linear (fish/100 m) and per area basis (fish per 100 m²).

Juvenile rainbow trout standing stock estimates were calculated for the two strata that could be sampled effectively using electrofishing, the 45 km of the Sutherland River extending downstream from Sutherland Lake to top of reach 1 of Bustard (1989), and the lower 7 km of Duncan Creek (see section 2.2). Point estimates of standing stock for each age class (SS_k) were calculated simply as the mean linear density among sites in the stratum times the stratum length.

A bootstrapping routine, incorporating both non-parametric and parametric components (Haddon 2001), was utilized to estimate 95% confidence limits for the standing stock estimates. The non-parametric component was included to address the potential problem of a non-normal or unknown distribution of abundance levels among sites in a reach, and relies on repeat sampling of the actual data rather than a theoretical probability distribution. Mean stratum density per km for each bootstrap iteration (i) was calculated as:

$$D_{i,k} = \frac{\sum_{j=1}^n \left(\frac{C_{1,i,j,k}}{L_{i,j}} \right) \times 1000 / n}{\hat{P}(cap)_{1,k}} \quad (5)$$

where:

- $D_{i,k}$ = mean density of age class k for iteration i
- n = number of sites in the stratum
- $C_{1,i,j,k}$ = first pass catch of age class k in the j th site selected by the bootstrap routine in iteration i
- $L_{i,j}$ = length of j th site (m) selected by the bootstrap routine in iteration i
- $\hat{P}(cap)_{1,k}$ = ML estimate of mean first-pass capture efficiency for the age class

To derive the 95% confidence limits, 2000 iterations of equation (5) were computed in Excel, with the bootstrap routine selecting sites with equal probability and replacement (Haddon 2001). To account for additional error associated with uncertainty in first-pass capture efficiency, in each iteration of the bootstrap routine values for $\hat{P}(cap)_{1,k}$ were stochastically generated using the random number generator routine in Excel, the ML estimate of mean first-pass capture efficiency for the age class, and the approximate standard error estimate for mean first-pass capture efficiency (see section 2.4.2). The standing stock of a given age class k for the stratum was estimated for each iteration as:

$$SS_{i,k} = D_{i,k} \times L_s \quad (6)$$

where L_s is the length of the stratum in km. Upper and lower 95% confidence limits for standing stocks were then estimated as the 2.5% and 97.5% percentiles of the 2000 bootstrap iterations. As a measure of precision, we calculated percent relative error, or the average half confidence interval as a proportion of the point estimate (Krebs 1999), for each of the bootstrapped confidence interval estimates.

Potential fish production from the lower 25 km of the Sutherland system could be estimated only from electrofishing data from open sites. Because it could not be assumed that capture efficiency was comparable to upstream reaches or to previous sampling, density estimates in this reach were not calculated.

Comparisons of juvenile rainbow trout abundance between September 2008 and September 1988 were made on the basis of estimated fish density for the two strata for which sampling by electrofishing was deemed effective: Duncan Creek and the Sutherland River mainstem above Reach 1 of Bustard (1989)(Figure 1). Age class-specific mean density estimates for 1988 were calculated from site density data in Bustard (1989), and compared to 1) reach density estimates based on all sites sampled in 2008, and 2) density estimates based only on site data collected from the same approximate stream locations utilized in 1988.

2.4.4 Predictions of maximum density

Ptolemy (1993) examined fish sampling data for eight salmonid species and found that nutrient levels in British Columbia streams were relatively good predictors of maximum salmon density at a given fish size. For systems in which suspended sediment levels are low, such as the Sutherland River system, the regression of maximum fish density FPU (fish/100 m²) on fish size $SIZE$, (g) and total alkalinity ALK (mg/L) is described by:

$$\log_{10} FPU = 1.58 - 0.97 \log_{10} SIZE + 0.45 \log_{10} ALK \quad (7)$$

(Ptolemy 1993). More recently, the relationship between maximum salmonid biomass among age classes $BIOMASS_{max}$ and total alkalinity has been described for BC coastal cutthroat trout (*Oncorhynchus clarki clarki*) streams by the simple power equation:

$$BIOMASS_{max} = 35 ALK^{0.67} \quad (8)$$

(Ptolemy 2005).

To enable model predictions of maximum salmonid density for the Sutherland River system, we first estimated total alkalinity during 2008 sampling based on observed measurements of conductivity CON (μS/cm). The following predictive equation was utilized:

$$ALK = 0.421 CON - 2.31 \quad (9)$$

(Ptolemy 1993). The only other historical conductivity/alkalinity data was collected during Environment Canada sampling of the Sutherland River on October 21, 1966 (Ron Ptolemy, MOE Victoria, data on file).

The total alkalinity estimate was utilized to make predictions of maximum density for a given age class and mean fish size. These were made using equations (7) and (8) above, respectively.

Model predictions were compared to empirical sampling data from 2008 and 1988 expressed in terms of 'biomass envelope' (Ptolemy 2005) lines on log-log plots of fish density versus fish size. Following Ptolemy (2005), the biomass envelope was estimated as the upper 95th percentile confidence limit of biomass/100 m² for the age class of highest biomass in the Sutherland River system. Data were pooled for Duncan Creek and the Sutherland River mainstem because of low sample size in Duncan Creek for 1988 in particular ($n = 3$; Bustard 1989).

3.0 RESULTS

3.1 Fish sampling

Fieldwork for this study was completed between September 2 and September 6, 2008. Discharge was visually estimated to be 1.5 m³ (50 cfs) on September 5 below Gravel Creek (Site DBS6 16.9, Figure 1), the lowest point in the system at which electrofishing sites could be enclosed by stop nets. Eighteen sites were sampled by electrofishing during the September 2-6 period ranged from 12-45 m in length (mean=22.1 m) and from 65-221 m² in area (mean=129 m²)(Appendix 1). Mean stream widths and thalweg depths among sites in the Duncan Creek (Figure 4), upper Sutherland River mainstem (Figure 2), and lower Sutherland River (Figure 5) sampling strata (see Section 2.2) were 3.9 m, 8.0 m, and 14.8 m and 34 cm, 56 cm, and 83 cm, respectively. Recorded stream temperatures ranged from 7.5°C-10.0°C during the course of sampling.

Rainbow trout were by far the most common fish species captured during sampling and dominated the catch at all sites (Appendix 2). Coho salmon, mountain whitefish, and prickly sculpin had a patchy distribution and were found only at very low levels of abundance (Appendix 2). Coho were found in the lower end of Duncan Creek (sites DU 1.03 and DU 1.25) and the first site downstream of Duncan Creek in the Sutherland mainstem (DBS3 27.9), as well as at a single location in the lower Sutherland River (S 14.7). Kokanee salmon spawners were present in relatively low densities throughout the Sutherland mainstem downstream of Duncan Creek. Sockeye salmon spawners were also observed, in the vicinity of Shass Creek. Both kokanee and sockeye were avoided when sampling.



Figure 4. Electrofishing site DU 1.25 in lower Duncan Creek.



Figure 5. Electrofishing site S 14.7 in the lower Sutherland River, which could not be effectively enclosed by stop nets.

Rainbow trout age classes 0+ through 3+ were identified for the Sutherland River system based on length-frequency and scale age data. When compared across the system, length-frequency data suggested four natural groupings of sites for estimating length-age cut-offs (used for assigning ages to fish captured in respective stream sections). These were Duncan Creek sites, those in the upper Sutherland mainstem upstream of Duncan Creek, sites between Duncan Creek and the top of the lower Sutherland stratum, and sites of the lower Sutherland River (reach 1 of Bustard 1989)(Table 1; Figures 6a-d). Length-age cutoffs were, in general, at lower fork lengths in upstream reaches (Table 1). Scale age data (Appendix 3) exhibited good agreement with length-age cutoffs estimated using length-frequency histograms. The scale age data did indicate, however, modest overlap of age-1+ and age-2+ parr lengths in the mid-Sutherland mainstem section, and of age-2+ and age-3+ parr lengths in the upper Sutherland section (Table 2).

Table 1. Length-age cut-offs used for assigning ages to sampled juvenile rainbow trout in the Sutherland River system, September 2008.

Section (see methods)	0+/1+	1+/2+	2+/3+	Comment
Duncan	65	102	136	Limited information about top end of 2+
Upper Sutherland	60	95	126	Based on scale information
Mid Sutherland	64	107	146	
Lower Sutherland	62	111	146	No 3+

Table 2. Range of fish sizes per age class, as determined by scale analysis.

Section (see methods)	Age-0+		Age-1+		Age-2+		Age-3+	
	min	max	min	max	min	max	min	max
Duncan	33	50	67	100	102	127	na	na
Upper Sutherland	41	49	65	93	95	134	114	140
Mid Sutherland	na	na	67	112	98	145	179	179
Lower Sutherland	40	42	67	110	na	na	na	na

Log-transformed fork length measures ($\log_{10} FL$) were good predictors of log-transformed weights ($\log_{10} Wt$) in simple linear regression ($R^2 = 0.99$), as expected (Le Cren 1951). The regression equation was:

$$\log_{10} Wt = 3.15(\log_{10} FL) - 5.26 \quad (10)$$

where Wt is measured in g and FL in mm. Equation (10) was the basis for estimating age class-specific biomass among sampling sites (see below).

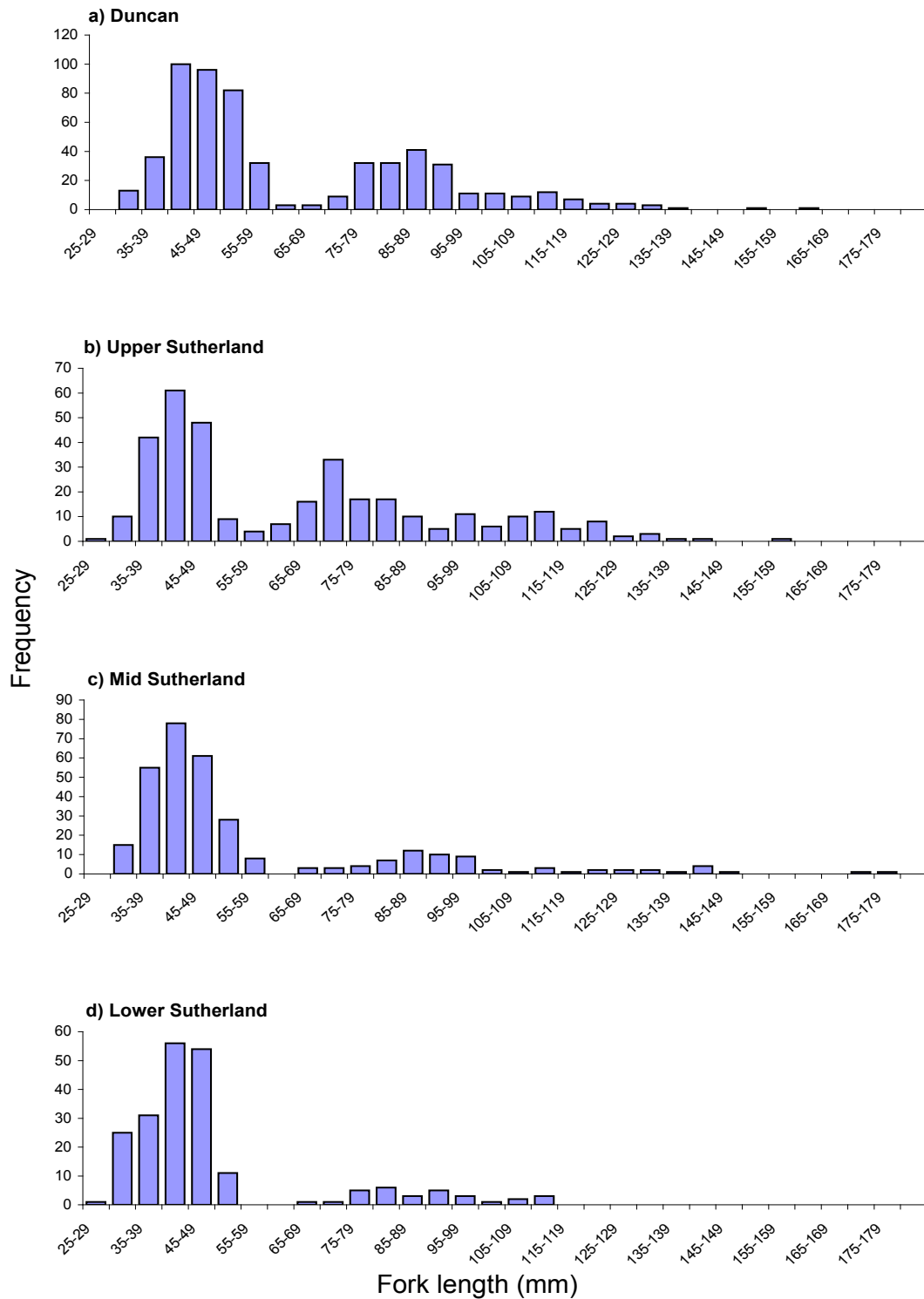


Figure 6. Length-frequency histograms for rainbow trout captured in a) Duncan Creek, b) the upper Sutherland River above Duncan Creek, c) the mid-portion of the Sutherland River between Duncan Creek and the lower Sutherland stratum (Reach 1 in Bustard 1989), and d) the lower Sutherland River.

Because sampling fish to determine sexual maturity (Figure 3) was destructive, relatively few were sacrificed from the Sutherland River system. Fish selected for assessing sexual maturity were also generally larger and therefore not a random sample. Strong patterns nonetheless were apparent in the sampling data. Among age1+ and older parr, fish that were mature precociously were all male (Table 3). Observed rates of maturity among sample age classes (Table 3) also increased with older age classes, with all sampled 3+ parr being sexually mature males¹. The lack of females and immature parr in the sample of age-3+ fish suggests that the majority of adfluvial rainbow trout leave the system for Babine Lake after their third year of life.

Table 3. Maturity of Sutherland system rainbow trout, by estimated age-class.

	Age-1+	Age-2+	Age-3+
Sample (<i>n</i>)	8	48	9
Number immature	6	30	0
Mature males	2	18	9
Mature females	0	0	0
Percent mature	25%	38%	100%

Genetic analysis of rainbow trout tissue samples, which were collected from a subset of the total number of sampling sites, was based on analysis of only six microsatellite loci and should therefore be considered imprecise (Koehler 2009). The preliminary analysis suggested that the rainbow trout sampled in the Sutherland River and Duncan Creek in 2008 may belong to more than one population or sub-population. The phylogenetic tree of genetic relations (Figure 7) suggests first that fry and parr from sites DBS7 35.6 and DBS6 16.9 along with fry from site S 41.6 form an important grouping (Koehler 2009) that occupies the core of the rainbow trout distribution in the Sutherland watershed. Second, parr from S 41.6 and the mixed sample (fry and parr tissue samples were combined) from S 45.1, the uppermost sampling site along the Sutherland mainstem, group together and are distinct from the first group. Fry from the uppermost sampling site in Duncan Creek group with the upper Sutherland fish, and parr from this site appear to be a third group.

3.2 Single-pass electrofishing capture probability

Maximum likelihood (ML) estimates of juvenile rainbow trout abundance at 3-pass electrofishing sites, used for calibrating first-pass catches by estimating first-pass capture probability $P(cap)_1$, were available from three sites distributed to distinctive stream sections: Duncan Creek (DBD1 3.03), the upper Sutherland River mainstem above

¹ Precociously mature fish were mature but not ripe, i.e. spawning would likely take place the following spring spawning period.

Duncan Creek (DBS7 35.6), and the Sutherland mainstem below Duncan Creek (DBS3 27.9)(Figure 1). Three-pass electrofishing data indicate consistently strong depletion

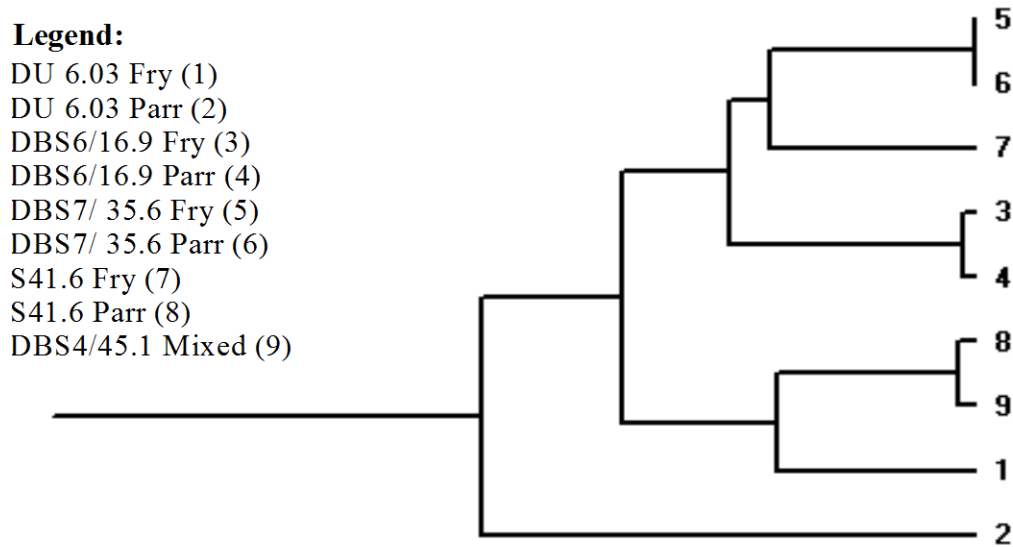


Figure 7. Phylogenetic tree showing genetic relationships among juvenile rainbow trout sampled in the Sutherland river system, as determined by molecular genetic analysis of 6 microsatellite loci. The length of the horizontal lines represents the level of genetic distinction among samples (Koehler 2009).

patterns across sites and age classes, resulting in relatively precise estimates of abundance N at calibration sites (Table 4). Across age classes, estimates of $P(cap)_1$ were relatively consistent among 3-pass sites (Table 4), ranging from 0.68-0.75, 0.60-0.80, and 0.60-0.83 for age classes 0+ to 2+, respectively. This was despite the relatively wide ranges of stream width, thalweg depth, substrate composition, and cover characteristics represented in the calibration sites (Appendix 1). First-pass capture probability could not be estimated for age-3+ rainbow trout because of an insufficient number of captures (Table 4).

Mean first-pass electrofishing capture probability $\hat{P}(cap)_1$, estimated using the method of maximum likelihood and calibration data from all sites, did not appear to be related to fish size at calibration sites. Estimates of $\hat{P}(cap)_1$ were very similar among age classes, and were 0.73, 0.73, and 0.76 for age classes 0+ through 2+, respectively. Precision of estimates of $\hat{P}(cap)_1$ was affected by the number of captures and was highest for age-0+ rainbow trout (the most frequently encountered), with the confidence

interval for $\hat{P}(cap)_1$ for this age class being less than half as broad as that for age-2+ fish (Table 5).

Table 4. Maximum likelihood estimates of rainbow trout abundance N at 3-pass removal electrofishing sites in the Sutherland River system, September 2008.

Site/Age class	Electrofishing catches by pass			<i>N</i>	SE(<i>N</i>)	95% confidence limits		First-pass
	1	2	3			LCL	UCL	<i>P</i> (cap)
DBD1 3.03								
Age-0+	66	18	9	96	2.61	93	102	0.68
Age-1+	30	4	3	37	0.81	37	39	0.80
Age-2+	15	3	0	18	0.27	18	19	0.83
Age-3+	1	0	0	-	-	-	-	-
DBS3 27.9								
rb0	66	13	7	88	1.66	86	91	0.75
rb1	10	6	0	17	1.01	16	19	0.60
rb2	5	3	0	8	0.72	8	10	0.60
rb3	0	0	0	-	-	-	-	-
S 35.6								
rb0	72	18	5	97	1.63	95	100	0.74
rb1	27	9	2	39	1.37	38	42	0.69
rb2	11	3	1	15	0.80	15	17	0.72
rb3	1	0	0	-	-	-	-	-

Table 5. Maximum likelihood estimates of mean first-pass capture efficiency relative to three-pass abundance estimates at calibration sites in the Sutherland River system, September 2008.

Age class	Mean first-pass		LCL	UCL
	capture efficiency	Approximate SE		
0+	0.73	0.026	0.68	0.78
1+	0.73	0.045	0.63	0.81
2+	0.76	0.064	0.61	0.87
3+	utilizes age-2+ estimate			

3.3 Juvenile rainbow trout abundance

Age-0+ rainbow trout densities varied widely among sampling sites ($CV = 0.76$) in the Sutherland River system, ranging from 46-1,142 fish/100 m (Appendix 4). Among sampling sites that could be enclosed by stop nets (upstream of Reach 1 of Bustard 1989), age-0+ density was negatively correlated with linear distance (in km, incorporated into site names) from Babine Lake ($r = 0.79$; Figure 8a). Rainbow trout fry densities were

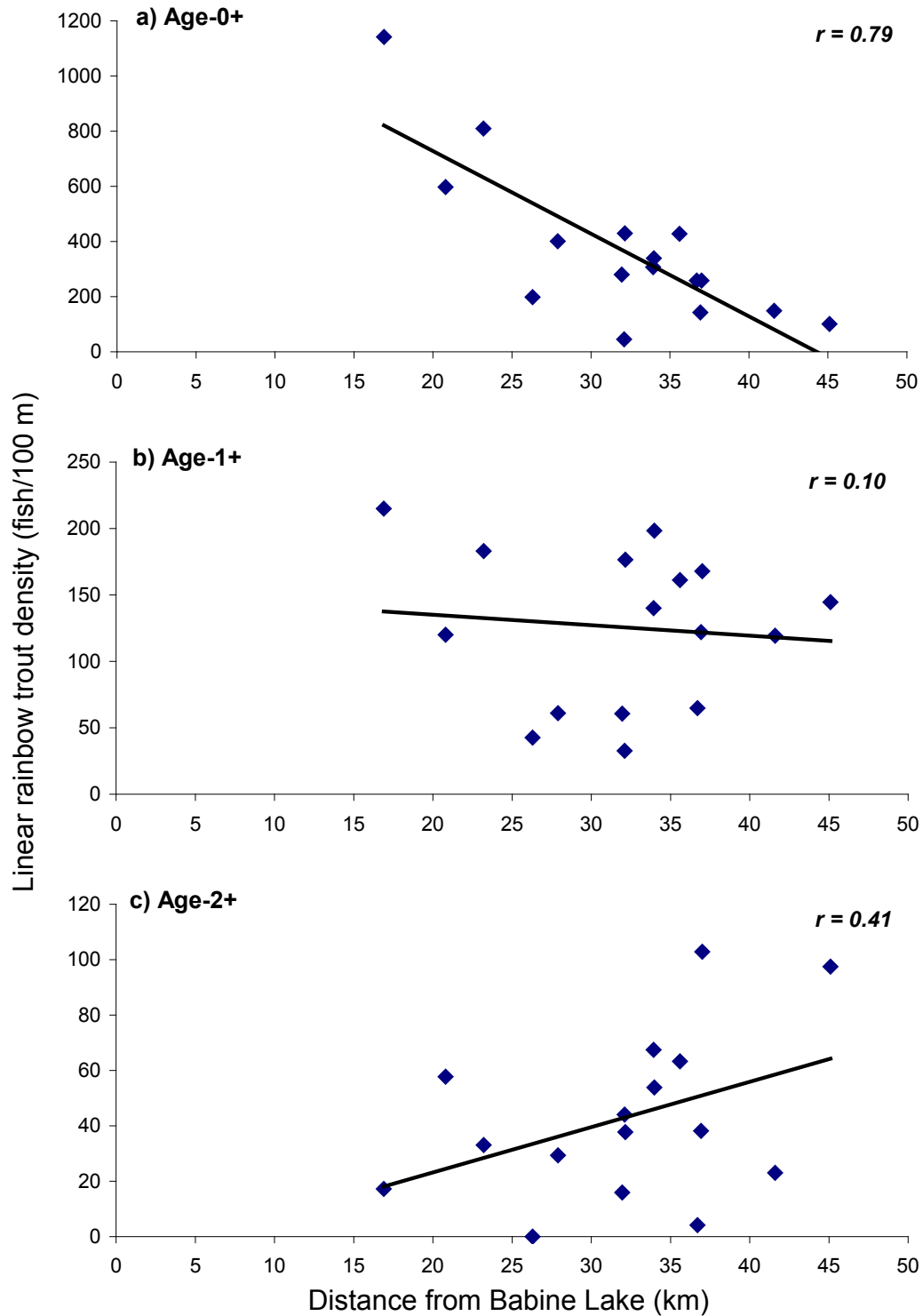


Figure 8. Correlation among sites of estimated fish density with distance of sampling site from Babine Lake, for a) age-0+, b) age-1+, and c) age-2+ rainbow trout.

highest (598-1,142 fish/100 m; Appendix 4) in the Sutherland River mainstem in the vicinity of Gravel Creek (Site DBS6 16.9; Figure 1) and at the next two sampling sites upstream (S 20.8, DBS2 23.2), indicating this stream section is important for rainbow trout spawners. Relatively high fry concentrations were also found in Duncan Creek sites other than the uppermost site (258-430 fish/100 m), and in the upper Sutherland River mainstem at sites DBS7 35.6 (428 fish/100 m) and DBS3 27.9 (401 fish/100 m). Fry densities from open sites in the lower Sutherland River mainstem (Reach 1 of Bustard 1989) were relatively lower (169-189 fish/100 m).

Age-0+ distribution patterns were not reflected by those of older age classes. Abundance estimates from open sites in the lower Sutherland were assumed to have potentially significant underestimation bias for more mobile older age classes. At locations that could be electrofished using enclosed sites, age-1+ densities exhibited lower variability among sites ($CV = 0.46$) and were not correlated with the site's distance from Babine Lake ($r = 0.10$; Figure 8b). Young rainbow trout have not been detected in Babine Lake (Griffiths 1968, as cited in Bustard 1989), suggesting that redistribution of fish or density-dependent, differential survival may occur along the length of the Sutherland River system to account for the shift in juvenile rainbow trout distribution. Age-2+ rainbow trout densities were more variable ($CV = 0.70$), and in contrast to age-0+ densities were positively correlated with the site's distance from Babine Lake ($r = 0.41$; Figure 8c). The distribution of age-3+ rainbow trout was the most patchy among age classes ($CV = 1.79$), with none being captured at more than half the number of sampling sites. Only three sites contained more than one age-3+ rainbow: DBS3 45.1 and S 37.0 in the upper Sutherland mainstem, and S 20.8 in the Sutherland mainstem downstream of Duncan Creek (Appendix 4).

Point estimates of mean juvenile rainbow trout linear density for the 43.5 km section of the Sutherland mainstem that could be electrofished effectively (above reach 1 in Bustard 1989) were 413, 125, 47, and 6.1 fish/100 m for age classes 0+ to 3+, respectively (Table 6). The point estimates for the 7.0 km of Duncan Creek thought to be accessible to adfluvial fish (Bustard 1989) were 293, 127, 36, and 1.2 fish/100 m for the same respective age classes. For the Duncan Creek stratum, precision among the density estimates, computed by bootstrapping the first-pass catch data along with the mean first-pass capture efficiency estimate for each age class, was greatest for the age-0+ population (% relative error² = 26%; 95% CI = 219-368) and declined for older age classes (Table 6). This reflects the effects in the bootstrap routine of both consistently high fry abundance in Duncan Creek and greater uncertainty in the estimates of $\hat{P}(cap)_i$ with increasing age category (Table 5). In the Sutherland mainstem stratum, however, the more precise estimate of $\hat{P}(cap)_i$ for age-0+ rainbow trout was offset by high spatial variability in abundance among sites (Appendix 4), and the most precise density estimate was for the age-1+ age class (% relative error = 32%; 95% CI = 87-167).

² Defined in Methods as the confidence interval divided by two and expressed as a percentage of the point estimate (Krebs 1999).

Rainbow trout standing stock estimates for the Duncan Creek and Sutherland mainstem sampling strata, and for both combined, were respectively: 180,000, 20,500, and 200,000 for age-0+; 54,300, 8,900, and 63,200 for age-1+; 20,400, 2,540, and 22,900 for age-2+; and 2,640, 87, and 2,720 for age-3+ (Table 6). The standing stock estimates for the Sutherland system (both strata combined) correspond to annual survival rates of 32% and 36% for the 0+-1+ and 1+-2+ transitions, respectively, assuming equilibrium population dynamics and negligible emigration prior to age-3. Overall, the most precise standing stock estimate, for the portion of the Sutherland system that could be sampled effectively, was that for age-1+ rainbow trout (% relative error = 29%; Table 6), followed by age-2+ (% relative error = 40%) and age-0+ (% relative error = 45%), and age-3+ (% relative error = 84%) age classes.

Standing stock estimates for the same strata based on 1988 sampling do not indicate a significantly larger population. These are higher for age-0+ rainbow trout, lower for age-1+, and comparable for age-2+ (Table 6), and all are within bootstrapped 95% confidence intervals derived from 2008 data. Age-3+ rainbow trout were not identified in 1988 (Bustard 1989). Comparing standing stock point estimates more directly, by basing them only on sites in the two sampling strata that were sampled during both periods, yields very close estimates for age-0+ and age-2+ age classes with a substantial discrepancy only for the estimated age-1+ standing stocks (75,400 in 2008 versus 48,900 in 1988; Table 6).

Table 6. Estimates of linear density and standing stock for juvenile rainbow trout in two strata of the Sutherland River system. Point estimates of 1988 standing stock are derived from density data in Bustard (1989).

Stratum/Age class	Density (fish/100 m)	Bootstrap lower 95% CL	Bootstrap upper 95% CL	Reach length (km)	Total standing stock	Bootstrap lower 95% CL	Bootstrap upper 95% CL	Percent relative error*
Upper Sutherland								
Age-0+	413	223	628	43.5	179,632	97,151	273,209	49%
Age-1+	125	87	167		54,261	37,644	72,476	32%
Age-2+	47	28	70		20,365	11,969	30,545	46%
Age-3+	6.1	1.3	12		2,636	584	5,155	87%
Duncan								
Age-0+	293	219	368	7	20,501	15,311	25,773	26%
Age-1+	127	84	172		8,896	5,871	12,026	35%
Age-2+	36	18	56		2,537	1,284	3,897	51%
Age-3+	1.2	0	2.9		87	2	200	115%
Combined standing stocks								
Age-0+	-	-	-	50.5	200,133	118,090	297,744	45%
Age-1+	-	-	-		63,156	45,384	81,512	29%
Age-2+	-	-	-		22,901	14,541	32,745	40%
Age-3+	-	-	-		2,722	622	5,181	84%
Combined 2008 standing stock point estimates from only those sites included in both 1988 and 2008 studies								
Age-0+	-	-	-	50.5	273,858	-	-	-
Age-1+	-	-	-		75,435	-	-	-
Age-2+	-	-	-		23,469	-	-	-
Combined 1988 standing stock point estimates from only those sites included in both 1988 and 2008 studies								
Age-0+	-	-	-	50.5	283,241	-	-	-
Age-1+	-	-	-		48,866	-	-	-
Age-2+	-	-	-		20,484	-	-	-

*Average half confidence interval divided by the point estimate (Krebs 1999)

3.4 Maximum density

Because of the relatively limited number of sampling sites in Duncan Creek, for 1988 sampling in particular ($n = 3$; Bustard 1989), fish density and conductivity data were pooled for Sutherland sampling strata prior to evaluating maximum rainbow trout density. Estimated mean conductivity for the Sutherland system in September 2008 was 122 $\mu\text{S}/\text{cm}$ ($\text{SE} = 11.4 \mu\text{S}/\text{cm}$), which corresponds to an estimated mean total alkalinity of 48.9 mg/L (see Section 2.4.5, Equation 9). The October 21, 1966 estimates of 210 $\mu\text{S}/\text{cm}$ conductivity and 107 mg/L total alkalinity (R. Ptolemy, BC Ministry of Environment Victoria, data on file) exhibit very poor agreement with our sampling data. Model predictions are based on concentrations at the ‘critical period streamflow’, defined

as the lowest monthly flow during the July 1-October 31 period. Because conductivity and total alkalinity are both inversely related to streamflow (Ptolemy 1993), and streamflow was elevated in September 2008 due to rainfall that had occurred during the previous week, we utilized the October 1966 estimate of total alkalinity in estimating maximum density.

The empirical ‘biomass envelope’ (Ptolemy 2005) is estimated as the upper 95% confidence limit of biomass/100 m² for the age class of highest biomass. For 1988 sampling data (Bustard 1989), the rainbow trout age class of maximum biomass and highest upper 95% confidence limit for biomass (Table 7) was that of age-1+ parr (mean among sites = 208 g/100 m²; 95% CL = 692 g/100 m²). By a narrow margin over age-2+ parr biomass, age-1+ parr biomass was also the highest for 2008 sampling data (mean among enclosed electrofishing sites = 144.1 g/100 m²; 95% CL = 315 g/100 m²). We therefore used the estimated biomass envelope for the 2008 age-1+ age class for comparison with model predictions (Figure 9a). It should be noted, however, that estimated biomass envelope for the age-2+ age class in 2008 was higher (mean = 143.7 g/100 m²; 95% CL = 379 g/100 m²).

Table 7. Estimates of juvenile rainbow trout biomass and maximum density for age-1+ rainbow trout parr, the age class of maximum biomass, in 1988 and 2008.

	Biomass (g/100 m ²)			Predicted max. density		
	Mean	Stdev	Biomass envelope	Age-1+ density at biomass envelope	Ptolemy 1993	CCT model
2008 sampling data	144	87	315	52	54	131
1988 sampling data	208	247	692	84	40	97

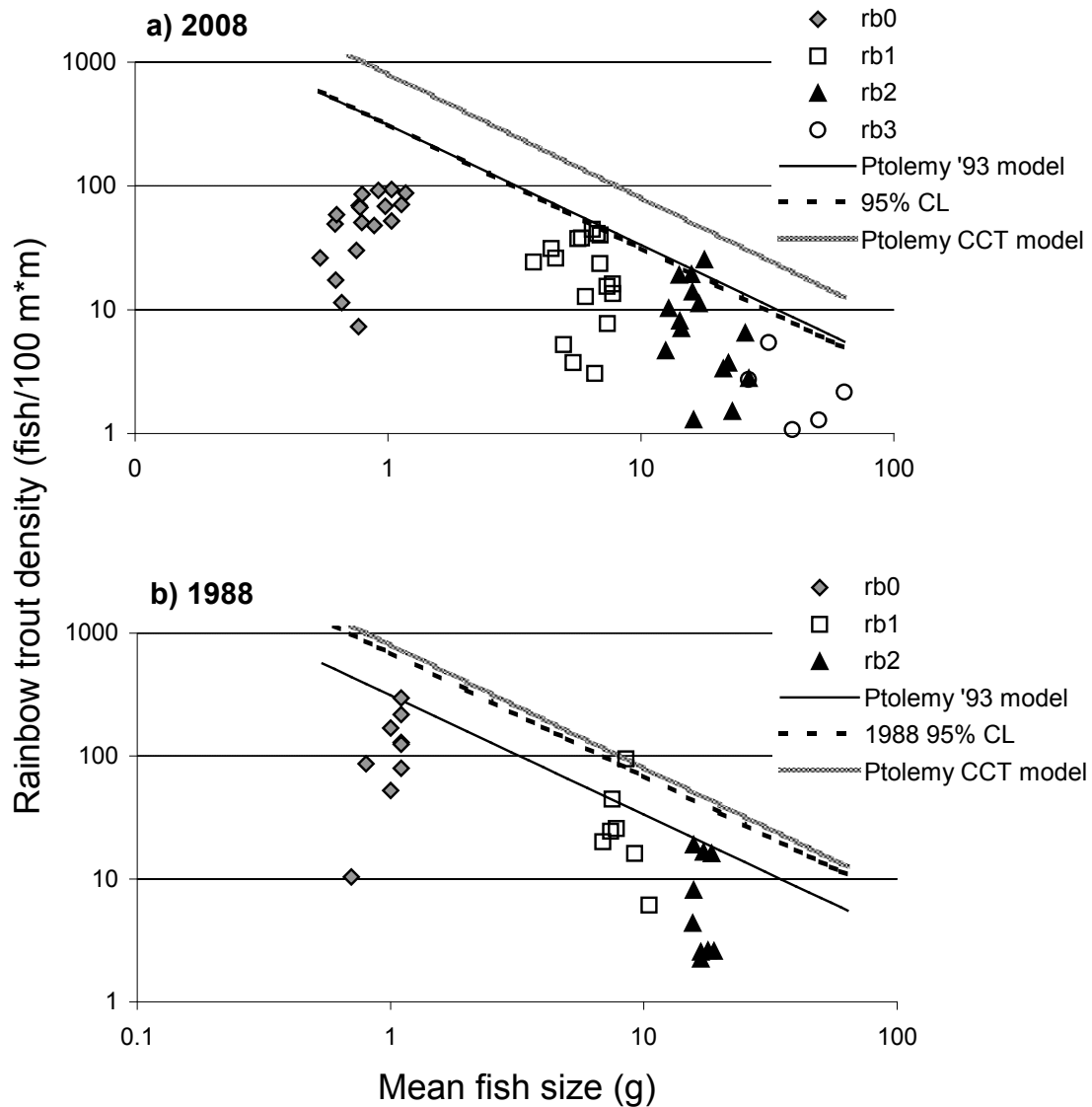


Figure 9. Comparisons of empirical estimates of maximum density in a) 2008 and b) 1988, estimates of which are derived from the upper 95% confidence limit for the age class of greatest biomass, with model predictions based on the Ptolemy (1993) general salmonid model and a model developed for coastal cutthroat streams (Ptolemy 2005).

Given that the empirical estimate of the biomass envelope for 1988 age-1+ parr sampling data is roughly double that for 2008 data, the comparisons of empirical estimates with model predictions of maximum density at an estimated mean total alkalinity of 107 mg/L (Table 7; Figures 9a-b) suggest a much more fully seeded rearing environment in 1988. At the mean fish size among sites for the age-1+ age class in

September 1988, the empirical estimate of maximum density, based on the estimated biomass envelope, exceeded predictions of the Ptolemy (1993) general salmonid model by 109% (Table 7; Figure 9b). The estimate was in better agreement with the Ptolemy (2005) coastal cutthroat trout (CCT) model prediction (14% lower; Table 7). In contrast, the empirical estimate of maximum density for the age-1+ age class in September 2008 exhibited good agreement with the general salmonid model prediction (4% lower; Table 7; Figure 9a), but was only 61% of the CCT model's prediction.

These large discrepancies between the estimated 1988 empirical biomass envelope and the 2008 biomass envelope estimate (and differential support for the two models' predictions), can be attributed to the exceptionally high biomass estimate of 805 g/100 m² for age-1+ at site S4 (Bustard 1989; site DBS4 45.1 in 2008). If the biomass estimate from this site is excluded from the analysis for 1988 sampling data, estimates of mean biomass among sites and the biomass envelope for the age-1+ age class (mean = 133 g/100 m²; 95% CL = 353 g/100 m²) are much more comparable to the 2008 estimates.

4.0 DISCUSSION

Rainbow trout dominate fluvial habitats in the Sutherland River system, with mountain whitefish, coho salmon, and cottids having a patchy distribution and occurring only at low abundances (Appendix 2). Coho salmon utilize habitats up to at least the lower end of Duncan Creek. Juvenile coho salmon were infrequent in electrofishing catches, but these may not be a good index of coho abundance given their known ability to utilize still water areas, off-channel habitats, and beaver dam impoundments (McPhail 2007), which are abundant in the Sutherland system and are not electrofished effectively.

Despite the broad diversity of stream sections sampled, including several pools of over 1 m maximum depth (Appendix 1), no rainbow trout older than age-3+ were captured during sampling in September 2008. Furthermore, all age-3+ sampled for sexual maturity were precocious males, and no sexually mature females were identified, suggesting that the population(s) inhabiting the mainstem Sutherland River system and lower Duncan Creek have a migratory life history. The relatively high standing stock of age-2+ parr suggests that the dominant age of emigration is age 3.

Assuming equilibrium dynamics, the approximate annual age-1+ to age-2+ survival estimates (based on observed standing stocks of different cohorts) of 36% for 2008 and 42% for 1988 sampling data are consistent with estimates from other populations of migratory salmonids, suggesting that significant migration prior to age-3 is not occurring. Annual survival rate for age-1+ and older Atlantic salmon parr has been estimated to be approximately 40% (Symons 1979; Randall et al. 1986), and Tautz et al. (1992) found that a mean annual survival of 48.8% per year was a good approximation of Keogh River juvenile steelhead survival patterns.

Bustard (1989, 1990) suggested that numerous beaver dams located on the Sutherland River mainstem upstream of Duncan Creek may limit access for migratory

rainbow trout spawners in spring, and identified beaver dam removal as a potential enhancement measure. Given that a substantial, native population of migratory rainbow trout already exists in the upper Sutherland River system, beaver dam removal is probably not required nor appropriate. Redds observed during September 2008, along with prior radio telemetry observations (Bustard 1990), confirm a distribution of spawning activity at least as far upstream as site DBS7 35.6 along the upper Sutherland River mainstem, and to a point located a short distance downstream of site DBD3 5.8 in lower Duncan Creek (Figure 1). Preliminary genetic analysis indicated that the fry captured at the upper Sutherland mainstem site S 41.6 were most closely related to fish captured at downstream locations, suggesting that spawning occurs this far upstream on at least some years (parr were more closely related to fish captured upstream at DBS4 45.1; Figure 7). It is unknown whether the apparently distinctive genetic samples from the upper Sutherland River (parr from S 41.6, fry and parr from DBS4 45.1) and the top end of Duncan Creek (DU 6.03) represent distinct population components of the Babine Lake adfluvial population, a mix of Babine Lake fish and fish utilizing lakes at the top of the Sutherland and Duncan drainages, or separate populations that contain no Babine Lake migrants. Clearly, the latter scenarios would compromise the evaluation of maximum salmonid densities from 1988 and 2008 sampling periods as indices of the populations' status. It is also quite possible that the putative population structure is an artifact of the low number of loci utilized in this preliminary genetic analysis; complete analysis of 14 loci utilized by Koehler (2010) with separation of fry and parr samples should be completed to clarify this issue.

In the stream sections that could be electrofished effectively during both 1988 and 2008 sampling periods (Duncan Creek and the upper Sutherland mainstem above Reach 1 in Bustard 1989), juvenile rainbow trout standing stocks were comparable or higher in 2008 (Table 6). The comparison of abundance between years was more reliable than indicated by the levels of precision among standing stock estimates (Table 6). This is because the high spatial variability in juvenile rainbow trout density among sites, a principal source of sampling variance in the standing stock estimates, is partially accounted for in our study design by utilizing the same approximate sampling locations in both years.

Comparable levels of juvenile rainbow trout abundance for the two sampling periods are consistent with two positive management scenarios. In the first, adult brood spawner abundance associated with age-classes 0+ to 2+ were similar for both study periods. Although brood spawner escapement estimates were not available to relate to 1988 juvenile sampling data, mark-recapture estimates of spawner escapement were approximately 500 in spring 1989 (Bustard 1990). In addition to counter-indicating negative population growth, a current spawner population size of this magnitude is probably not of major short-term conservation concern (Boyce 1992; Nunney and Campbell 1993; McElhany et al 2000). Populations of large, piscivorous rainbow trout in British Columbia are all relatively small (Bustard 1990; Keeley et al. 2005; Hagen et al. 2010), and it is possible that none (with the possible exception of the Gerrard rainbow of Kootenay Lake; Hagen et al. 2010), have a spawner population size that is consistently greater than 1000 annually. Conservative management of piscivorous rainbow trout stocks should therefore be standard practice.

Similar population sizes may instead be due to density-dependent population regulation. For salmonid populations exhibiting an extended freshwater residency early in life, density-dependent mortality is thought to be prevalent in rearing habitats at the core of their distributions (Larkin 1956; Chapman 1966; Elliott 1987), and to be a regulator of population size. Strong compensatory mortality at high densities may operate most strongly in the period immediately following emergence (Elliott 1987), or may primarily effect age-0+ to age-1+ survival (Ward and Slaney 1993).

Assuming three years' stream residency, age-2+ standing stock estimates for the two sampling periods integrate the effects of density-dependent survival across the stream residency period and may therefore be the best basis for comparison (Table 6). In this second scenario, the similar age-2+ standing stock estimates indicate that spawner abundance and egg deposition were adequate to seed available rearing habitats. Given that density-dependent regulation of juvenile trout abundance typically operates over a broad range of potential spawner escapement levels (Elliott 1987; Ward and Slaney 1993), this scenario is not dependent on the coincidence of similar egg deposition levels to explain similar levels of juvenile abundance and is therefore more likely. Confidence in this conclusion, however, must be considered to be relatively low given that only two sampling periods can be compared, but will increase significantly if supported by additional years' sampling data. The alternative, egg deposition levels that are inadequate to seed available rearing space, would potentially be reflected in more variable parr abundance that was proportional to brood fry abundance across a multi-year time series.

Lacking information on spawner abundance, juvenile abundance levels that saturate available rearing habitats, as indicated by asymptotes of abundance exhibited over several sampling periods, are natural management reference points that can be utilized to signal the need for more stringent conservation management. Factors influencing a population's status are likely to operate on longer time scales than can be picked up by a short period of sampling over consecutive years. Correlation in abundance estimates among successive years is also a concern (Hilborn and Walters 1992). The comparison in this study of juvenile abundance estimates with past sampling data provides an important illustration of the value of inventory data, provided that it is collected in a rigorous manner that allows unbiased and reliable estimates of juvenile salmonid abundance to be made. As a cost-effective alternative when annual population monitoring is not feasible, the establishment of a baseline of abundance estimates, to which future comparisons (separated in time) can be made, should be considered as a potential approach for other fish populations in northern BC as well.

Bustard (1990) suggested that the estimated 10% exploitation rate of spring 1989, due to the Yekooche gillnet fishery at the mouth of the Sutherland River, may have been a substantial underestimate of typical annual fishing mortality at that time. Even if recent gillnetting effort and harvest are thought to be reduced from historical levels (Mark West, Burns Lake Conservation Officer Service, pers comm.), the fact that the fishery is not currently monitored or regulated poses a challenge to managers seeking the long-term conservation of this unique population. If the fishery remains unmonitored and unregulated, a commitment to periodic monitoring in the Sutherland River system is

probably required in order to ensure that a decline below levels necessary to seed available rearing habitats does not occur.

Results of this study suggest a number of concerns with our second approach to assessing the status of Sutherland River system rainbow trout: comparisons of maximum density with model predictions. Perhaps most seriously, estimates of maximum density (Table 7), based on estimates of the upper 95% confidence limit for biomass/100 m², were unreliable indicators of overall population abundance for the two sampling periods. To summarize, despite lower age-1+ parr abundance overall, the maximum density estimate for 1988 was more than two times greater than for 2008 (Table 7), which was primarily due to the influence of a single site. Estimation of habitat saturation levels based on reach averages, rather than maximum density estimates associated with 95% confidence limits for biomass, would probably improve reliability of the estimates as well as being more applicable to a broad range of management questions. Relative to biomass confidence limits, in most instances reach averages would be less sensitive to sample size and assumptions about the underlying statistical distribution for abundance. Little appears to be known about the spatial behaviour within rainbow trout populations at varying levels of abundance. Without this knowledge, the theoretical basis for the use of maximum density, rather than reach averages, is also unclear. It is conceivable that the highest quality habitats in core rearing areas are filled to saturation first, and exhibit less variation among years, while full utilization of marginal habitats may not occur unless total abundance is high enough to force emigration from the best areas.

The exceptional influence of site DBS4 45.1 on estimates of maximum density for the 1988 sampling period should be noted in evaluating observed abundance relative to model predictions. With the exclusion of sampling data from this site (data point of highest elevation relative to predictions of maximum density in Figure 9b), a visual inspection of the estimated densities among sites relative to model predictions suggests that the general salmonid model (Ptolemy 1993) is a good fit to the observations from both sample periods (Figures 9a, 9b). The CCT model is a poor fit to these data, suggesting either it is inappropriate for use in the Sutherland system, or that observed densities do not saturate the available rearing space. If site DBS4 45.1 is included along with the others, the CCT model predictions are supported, and the status of the population sampled during that period appears different relative to 2008.

Given that the geographic scope of sampling used to develop the CCT predictive model is necessarily focused on coastal areas that may have unique population dynamics relative to interior populations, the general salmonid model predictions are probably the most appropriate for evaluating sampling data from the Sutherland system. The good fit of predictions to the observed density data is consistent with the notion that observed densities are currently seeding available rearing habitats, and the population's conservation status has not been seriously degraded in the two decades following the last monitoring study. We caution, again, that confidence in this assessment will improve with additional years' sampling data.

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APPENDIX 1. Physical habitat attributes at sampling sites.

Stream	Site	Distance from lake	Date	Position (NAD 27)	Temp	Cond	%ri	%ru	Mean stream width	Channel width	Mean site length	Site area	Max site depth	Mean thalweg depth
Sutherland	DBS4 45.1	45.1	3-Sep	10 U 394780 6012999	8		50	50	3.9	4.0	19.0	73.2	50	35
Sutherland	S 41.6	41.6	4-Sep	10 U 392096 6015211			10	90	4.9	6	23.0	113.5	102	65
Sutherland	S 37.0	37.0	4-Sep	10 U 388364 6017923	8	70	30	70	5.4	9	18.0	96.9	55	43
Sutherland	DBS7 35.6	35.6	6-Sep	10 U 387340 6018991	9.5		15	85	6.2	10	23.0	141.8	68	47
Sutherland	S 32.1	32.1	4-Sep	10 U 383286 6019483	9.5		9	91	6.3	8	21.0	131.3	71	58
Sutherland	DBS3 27.9	27.9	6-Sep	10 U 380714 6022895			90	10	7.9	na	22.5	177.8	110	72
Sutherland	S 26.3	26.3	4-Sep	10 U 379698 6024151	8.5	120	20	80	11.4	14	19.3	220.7	96	74
Sutherland	DBS2 23.2	23.2	5-Sep	10 U 377413 6026304	9.5		15	85	11.8	15.5	12.0	142.0	110	72
Sutherland	S 20.8	20.8	5-Sep	10 U 375939 6028321	9	120	10	90	8.9	12	13.7	122.2	62	47
Sutherland	DBS6 16.9	16.9	5-Sep	10 U 373205 6031299	10		60	40	13.3	19	15.3	204.4	56	42
Sutherland	S 14.7	14.7	5-Sep	10 U 371380 6032480	10		5	95	14.8	17	12.1	179.6	105	83
Sutherland	S 7.05	7.05	5-Sep	10 U 365029 6036872	10	130	100	0	3.7	16	26.0	97.1	40	22
Duncan	DU 6.03	36.9	2-Sep	10 U 379663 6014790			35	65	2.7	5	45.0	123.0	31	23
Duncan	DBD3 5.8	36.7	2-Sep	10 U 379795 6014979	10		65	35	2.8	10	31.8	87.5	54	33
Duncan	DU 3.08	34.0	3-Sep	10 U 381639 6017195			25	75	4.8	6	27.0	129.6	84	42
Duncan	DBD1 3.03	33.9	3-Sep	10 U 381660 6017200	7.5	150	10	90	3.5	7	29.4	102.9	58	43
Duncan	DU 1.25	32.2	4-Sep	10 U 382440 6018907			25	75	4.7	6	14.0	65.3	35	29
Duncan	DU 1.05	32.0	4-Sep	10 U 382681 6019129	9.5	140	40	60	4.8	10	24.9	118.7	63	37

Discharge visually estimated to be 1.5 m³, 50 cfs on September 5 below Gravel Creek

APPENDIX 1, continued.

Stream	Site	Substrate		%Boulder	%Cobble	%Gravel	%Fines	Cover	m ² LWD	%DW	%Turb	%UC	%OH	%Veg
		D90	D50					%Sub						
Sutherland	DBS4 45.1	27	9	15	35	20	30	20	4	5	60	10	5	0
Sutherland	S 41.6	6	4	0	0	60	40	0	0	10	2	30	15	0
Sutherland	S 37.0	40	15	10	30	30	30	30	13	15	20	30	10	10
Sutherland	DBS7 35.6	20	5	2	23	60	15	10	5	5	10	30	15	0
Sutherland	S 32.1	0.5	0.2	0	0	10	90	0	10	15	10	0	20	0
Sutherland	DBS3 27.9	10	4	0	5	45	50	0	9	50	10	45	0	25
Sutherland	S 26.3	4	1	0	0	25	75	0	4	45	0	30	10	10
Sutherland	DBS2 23.2	5	2	0	0	80	20	0	5	15	20	10	10	0
Sutherland	S 20.8	12	6	0	30	50	20	0	0	30	15	20	5	0
Sutherland	DBS6 16.9	23	11	5	75	20	0	50	1	0	45	0	2	2
Sutherland	S 14.7	6	2	0	0	40	60	0	19	50	0	15	0	5
Sutherland	S 7.05	5	2	0	0	65	35	0	0	0	20	0	0	0
Duncan	DU 6.03	15	3	2	25	38	35	10	8	0	5	5	15	20
Duncan	DBD3 5.8	6	2	0	25	40	35	0	9	0	0	20	15	15
Duncan	DU 3.08	12	5	0	45	40	15	15	15	15	10	20	20	5
Duncan	DBD1 3.03	12	5	0	15	45	40	0	3	30	0	40	0	50
Duncan	DU 1.25	6	3	0	5	70	25	5	2.5	0	15	0	30	0
Duncan	DU 1.05	5	2	0	0	35	65	0	22	10	0	48	20	0

APPENDIX 2. First-pass electrofishing catch data.

Site	Site length (m)	Site area (m ²)	Uncalibrated first-pass catches								Mean rainbow trout size (g)			
			rb0	rb1	rb2	rb3	co	cott	ko	mw	rb0	rb1	rb2	rb3
DBS4 45.1	19.0	73.2	14	20	14	3	0	0	0	0	0.54	5.64	17.8	32.0
S 41.6	23.0	113.5	25	20	4	0	0	0	0	0	0.75	3.76	12.5	-
S 37.0	18.0	96.9	34	22	14	2	0	0	0	0	0.88	4.42	14.1	26.6
DBS7 35.6	23.0	141.8	72	27	11	1	0	0	0	0	0.77	4.60	12.9	31.3
S 32.1	21.0	131.3	7	5	7	0	0	1	0	2	0.77	4.94	14.4	-
DBS3 27.9	22.5	177.8	66	10	5	0	5	0	0	3	0.79	7.38	22.2	-
S 26.3	19.3	220.7	28	6	0	0	0	0	2	0	0.62	5.40	-	-
DBS2 23.2	12.0	142.0	71	16	3	0	0	1	0	1	0.97	7.34	26.7	-
S 20.8	13.7	122.2	60	12	6	2	0	0	0	0	0.77	7.73	25.8	63.8
DBS6 16.9	15.3	204.4	128	24	2	0	0	1	0	0	0.79	7.71	16.1	-
S 14.7	12.1	179.6	15	4	0	0	3	1	0	0	0.66	6.57	-	-
S 7.05	26.0	97.1	35	0	0	0	0	0	0	1	0.62	-	-	-
DU 6.03	45.0	123.0	47	40	13	1	0	0	0	0	1.03	6.43	16.0	39.7
DBD3 5.8	31.8	87.5	60	15	1	0	0	0	0	0	1.03	6.88	23.0	-
DU 3.08	27.0	129.6	67	39	11	0	0	0	0	0	1.13	6.71	17.0	-
DBD1 3.03	29.4	102.9	66	30	15	1	0	0	0	0	1.17	6.91	15.8	50.5
DU 1.25	14.0	65.3	44	18	4	0	4	0	0	0	0.91	5.83	14.2	-
DU 1.05	24.9	118.7	51	11	3	0	2	0	0	1	0.63	6.03	21.2	-

APPENDIX 3. Rainbow trout scale age data, September 2008.

Slide #	Stream section	Scale #	Fork length	Estimated age
D3	Duncan	10	33	0+
D1-25	Duncan	6	42	0+
D3	Duncan	19	50	0+
D1-25	Duncan	2	67	1+
D3	Duncan	3	69	1+
D3	Duncan	1	70	1+
D3	Duncan	17	72	1+
D3	Duncan	18	72	1+
D3	Duncan	12	77	1+
D3	Duncan	7	81	1+
D1	Duncan	3	84	1+
D1-05	Duncan	1	85	1+
D3	Duncan	5	85	1+
D3	Duncan	15	87	1+
D1-25	Duncan	4	89	1+
D3	Duncan	20	91	1+
D3	Duncan	6	92	1+
D3	Duncan	9	94	1+
D3	Duncan	2	98	1+
D3	Duncan	13	100	1+
D1	Duncan	9	102	2+
D1	Duncan	7	103	2+
D1-25	Duncan	1	104	2+
D1	Duncan	8	108	2+
D3	Duncan	21	108	2+
D1	Duncan	5	112	2+
D1-25	Duncan	3	116	2+
D3	Duncan	11	118	2+
D1	Duncan	6	120	2+
D1-05	Duncan	5	120	2+
D3	Duncan	14	121	2+
D1-05	Duncan	3	125	2+
D1-05	Duncan	6	126	2+
D1	Duncan	1	127	2+
S16-9	Lower Sutherland	3	40	0+
S16-9	Lower Sutherland	6	42	0+
S16-9	Lower Sutherland	5	68	1+
S16-9	Lower Sutherland	4	79	1+
S14-7	Lower Sutherland	2	92	1+

APPENDIX 3, continued

Slide #	Stream section	Scale #	Fork length	Estimated age
S16-9	Lower Sutherland	1	94	1+
S16-9	Lower Sutherland	8	102	1+
S16-9	Lower Sutherland	9	105	1+
S16-9	Lower Sutherland	7	110	1+
S27-9	Mid Sutherland below Duncan	10	67	1+
S26-3	Mid Sutherland below Duncan	3	69	1+
S23-2	Mid Sutherland below Duncan	1	72	1+
S27-9	Mid Sutherland below Duncan	2	80	1+
S26-3	Mid Sutherland below Duncan	1	87	1+
S20-8	Mid Sutherland below Duncan	2	92	1+
S27-9	Mid Sutherland below Duncan	1	94	1+
S27-9	Mid Sutherland below Duncan	12	98	2+
S20-8	Mid Sutherland below Duncan	3	102	1+
S27-9	Mid Sutherland below Duncan	9	107	2+
S20-8	Mid Sutherland below Duncan	4	112	1+
S27-9	Mid Sutherland below Duncan	4	114	2+
S27-9	Mid Sutherland below Duncan	3	118	2+
S27-9	Mid Sutherland below Duncan	5	123	2+
S27-9	Mid Sutherland below Duncan	7	123	2+
S27-9	Mid Sutherland below Duncan	11	134	2+
S23-2	Mid Sutherland below Duncan	2	137	2+
S27-9	Mid Sutherland below Duncan	6	143	2+
S23-2	Mid Sutherland below Duncan	3	145	2+
S20-3	Mid Sutherland below Duncan	1	179	3+
S32-1	Upper Sutherland above Duncan	4	41	0+
S32-1	Upper Sutherland above Duncan	3	49	0+
S37-0	Upper Sutherland above Duncan	4	65	1+
S4	Upper Sutherland above Duncan	12	68	1+
S37-0	Upper Sutherland above Duncan	1	69	1+
S37-0	Upper Sutherland above Duncan	9	69	1+
S35-6	Upper Sutherland above Duncan	6	74	1+
S37-0	Upper Sutherland above Duncan	8	80	1+
S4	Upper Sutherland above Duncan	3	93	1+
S4	Upper Sutherland above Duncan	13	93	1+
S35-6	Upper Sutherland above Duncan	4	95	2+
S35-6	Upper Sutherland above Duncan	10	96	2+
S37-0	Upper Sutherland above Duncan	6	97	2+
S35-6	Upper Sutherland above Duncan	12	98	2+
S41-6	Upper Sutherland above Duncan	2	98	2+

APPENDIX 3, continued

Slide #	Stream section	Scale #	Fork length	Estimated age
S4	Upper Sutherland above Duncan	5	99	2+
S35-6	Upper Sutherland above Duncan	8	102	2+
S35-6	Upper Sutherland above Duncan	11	102	2+
S37-0	Upper Sutherland above Duncan	2	102	2+
S35-6	Upper Sutherland above Duncan	3	103	2+
S35-6	Upper Sutherland above Duncan	5	105	2+
S37-0	Upper Sutherland above Duncan	7	105	2+
S41-6	Upper Sutherland above Duncan	6	109	2+
S35-6	Upper Sutherland above Duncan	2	110	2+
S37-0	Upper Sutherland above Duncan	5	113	2+
S41-6	Upper Sutherland above Duncan	5	113	2+
S35-6	Upper Sutherland above Duncan	9	114	2+
S37-0	Upper Sutherland above Duncan	11	114	3+
S32-1	Upper Sutherland above Duncan	1	116	2+
S4	Upper Sutherland above Duncan	1	118	2+
S4	Upper Sutherland above Duncan	8	118	2+
S4	Upper Sutherland above Duncan	10	118	2+
S4	Upper Sutherland above Duncan	11	118	2+
S32-1	Upper Sutherland above Duncan	5	122	2+
S4	Upper Sutherland above Duncan	14	122	2+
S35-6	Upper Sutherland above Duncan	7	123	2+
S37-0	Upper Sutherland above Duncan	3	125	3+
S4	Upper Sutherland above Duncan	9	125	2+
S37-0	Upper Sutherland above Duncan	10	130	3+
S4	Upper Sutherland above Duncan	2	130	2+
S4	Upper Sutherland above Duncan	7	134	2+
S35-6	Upper Sutherland above Duncan	1	140	3+

APPENDIX 4. Site-level rainbow trout density estimates.

Site	Maximum likelihood estimates of rainbow trout abundance												Site density (fish/100 m)				FPU (fish/100 m ²)			
	rb0	LCL	UCL	rb1	LCL	UCL	rb2	LCL	UCL	rb3	LCL	UCL	rb0	rb1	rb2	rb3	rb0	rb1	rb2	rb3
DBS4 45.1	19	15	25	27	22	36	19	14	26	4	3	8	101	145	97	21	26	38	25	5.4
S 41.6	34	28	42	27	22	36	5	4	9	0	0	0	149	119	23	0	30	24	4.7	0
S 37.0	47	39	56	30	24	39	19	14	26	3	2	6	258	168	103	15	48	31	19	2.73
DBS7 35.6	98	86	113	37	30	47	15	11	21	1	1	4	428	161	63	6	69	26	10	0.93
S 32.1	10	7	14	7	5	11	9	7	14	0	0	0	46	33	44	0	7.3	5.2	7.1	0
DBS3 27.9	90	78	104	14	10	20	7	5	11	0	0	0	401	61	29	0	51	7.7	3.7	0
S 26.3	38	31	47	8	6	13	0	0	0	0	0	0	198	43	0	0	17	3.7	0	0
DBS2 23.2	97	85	112	22	17	29	4	3	7	0	0	0	809	183	33	0	68	15	2.8	0
S 20.8	82	71	95	16	13	23	8	6	12	3	2	6	598	120	58	19	67	13	6.5	2.16
DBS6 16.9	175	157	197	33	27	42	3	2	6	0	0	0	1142	215	17	0	86	16	1.3	0
S 14.7	21	16	27	5	4	10	0	0	0	0	0	0	169	45	0	0	11	3.1	0	0
S 7.05	48	40	58	0	0	0	0	0	0	0	0	0	184	0	0	0	49	0	0	0
DU 6.03	64	55	77	55	46	68	17	13	24	1	1	4	143	122	38	3	52	45	14	1.08
DBD3 5.8	82	72	96	21	16	28	1	1	4	0	0	0	258	65	4	0	94	24	1.5	0
DU 3.08	92	80	106	54	44	67	15	11	21	0	0	0	339	198	54	0	71	41	11	0
DBD1 3.03	90	78	104	41	33	52	20	15	27	1	1	4	307	140	67	4	88	40	19	1.29
DU 1.25	60	52	72	25	19	30	5	4	9	0	0	0	430	177	38	0	92	38	8.1	0
DU 1.05	70	60	82	15	11	20	4	3	7	0	0	0	280	61	16	0	59	13	3.3	0