DISTRIBUTION OF ADFLUVIAL BULL TROUT PRODUCTION IN TRIBUTARIES OF THE ARROW LAKES RESERVOIR AND THE FEASIBILTY OF MONITORING JUVENILE AND ADULT ABUNDANCE

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June, 2007

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EXECUTIVE SUMMARY

The Columbia Basin Fish and Wildlife Compensation Program (CBFWCP) and BC Hydro Columbia Basin Generation are interested in developing a program to monitor adfluvial bull trout (*Salvelinus confluentus*) populations in the Arrow Lakes Reservoir (ALR) in response to reservoir fertilization and other compensation initiatives. As a first step towards this goal, better information is needed about which tributaries support adfluvial bull trout populations in the Arrow Lakes Reservoir (ALR), and the relative contribution of these tributaries to reservoir recruitment. Over a three-year period from 2004-2006 we evaluated night snorkeler counts and redd surveys as tools for monitoring the abundance of juvenile and adult bull trout, respectively, in ALR tributaries, and attempted to identify tributary reaches in the ALR that contributed significantly to overall production. We also included rainbow trout (*Oncorhynchus mykiss*) in our evaluation of snorkeler counts, and as part of our distribution and abundance study.

Mark-recapture data indicated that nighttime snorkeler counts, when calibrated, can provide accurate and reasonably precise estimates of bull trout and rainbow trout parr abundance in non-glacial tributaries. Snorkeling efficiency was relatively high for both bull trout (q = 49%; 95% confidence interval = 40%-57%) and rainbow trout (q = 72%; 95% CI = 61%-83%). Snorkeling efficiency was size-dependent for rainbow trout parr, but not for bull trout. When conducted later in the fall after glacial run-off had subsided, snorkeler counts also appeared well suited for sampling fish abundance in the considerably larger, glacial mainstems of the Incomappleaux and Illecillewaet rivers, although we did not investigate snorkeling efficiency in these reaches directly.

The evaluation of the bull trout redd counts as a monitoring tool was hampered during 2004 and 2005 by unusually high stream flows. In 2006 we were able to conduct complete redd counts in all study tributaries, including the glacial Incomappleaux and Illecillewaet systems. Together with approximations of juvenile standing stock derived from the snorkeling surveys, these estimates of spawner abundance allowed for the first time an analysis of the relative contribution of individual tributaries to overall spawning

abundance and recruitment. The Illecillewaet River system is clearly the most important bull trout spawning and rearing tributary in the ALR, accounting for nearly half of the redds observed in 2006 (449 of 953). The Incomappleaux River accounted for an additional 165 redd, and together these two large, glacial streams and their tributaries account for 64% of the total number of redds observed. The remaining, non-glacial tributaries in the study, Halfway, MacDonald, Caribou, and Kuskanax contributed 15%, 12%, 5%, and 4%, respectively, to the total redd count.

Late summer snorkeler counts indicated that the distribution of rearing adfluvial bull trout in non-glacial tributaries was much less extensive than previously assumed. Consistent spatial patterns in the distribution and age-class structure of bull trout and rainbow trout suggested resource partitioning within the watershed. The highest densities (12.6-101 parr/100 m or 1.1-10.2 parr/100 m^2) of bull trout parr were always associated with reaches separated from the reservoir either by major obstructions or lengthy sections of high gradient channel. Stream temperatures in these reaches were also relatively low. Bull trout were largely absent from lower reaches near the reservoir, which were dominated by rainbow trout, often at high densities (5.5-21.0 parr/100 m^2). Bull trout parr were mostly (>90%) age-1+ or age-2+, suggesting a tributary rearing phase of 2-3 years prior to reservoir entry. Rainbow trout that coexisted with bull trout in upper reaches had a diverse age class structure (age-1+ to 4+), suggesting a resident life history, whereas the higher numbers of primarily younger rainbow trout (age-1+ and 2+) in the lower reaches suggested an adfluvial life history.

Warmer stream temperatures and competition with rainbow trout are possible explanations for the lack of bull trout parr in lower reaches. This challenges the previous assumption that dam construction and the resultant flooding of habitat in lower tributary reaches substantially reduced rearing capacity for adfluvial bull trout. Migration barriers in Illecillewaet and Halfway rivers were removed in past attempts to compensate for the loss of tributaries north of Revelstoke Dam. Habitat upstream of these reaches accounted for 62% of the total redd count in 2006.

Rearing areas for bull trout in non-glacial ALR tributaries are highly constrained by migration barriers, water temperatures and competition from rainbow trout. These areas may be highly vulnerable to temperature increases resulting from forest harvesting and climate change, and to disturbances from run-of-the-river hydroelectric projects and other industrial activities. In the Incomappleaux system, most bull trout spawning tributaries showed signs of serious channel instability that was often associated with extensive streamside logging, and poor bridge and culvert road construction. In some cases, channel movement likely prevented adults from accessing spawning habitat. Managing and restricting resource development in bull trout spawning and rearing areas should be given high priority.

Recommendations for a long term monitoring program

- Inclusion of both juvenile snorkeler counts and adult redd counts as part of a long-term monitoring program would provide valuable stock and recruitment data for bull trout in the ALR. However, if resources were too few to collect both types of data, redd counts should take priority. Redd counts represent a direct index of the status of the spawner stock in the ALR and reflect reservoir-specific influences, and they can be related to the annual reservoir creel survey to assess exploitation rates and provide conservation guidelines. Consideration should also be given to conducting redd surveys in tributaries not surveyed in this study that may support bull trout spawning, with priority given to relatively uncommon populations of the southern genotype. Juvenile abundance data may be most useful for identifying factors that limit the distribution of bull trout rearing within the ALR catchment.
- Temperature loggers should be deployed in reaches sampled during 2004-2006, so that maximum summer temperature data can be acquired and related to mean bull trout and rainbow trout parr densities.
- The possibly of ALR bull trout exhibiting early life-history strategies other than indicated by our data (i.e, 2-3 years of tributary rearing prior to emigration to the reservoir) should be investigated as this has implications with respect to whether tributary rearing space represents a bottleneck to adult abundance in the reservoir. As a first step, we recommend that otolith microchemistry be conducted on the existing

library of adult bull trout otoliths collected from the reservoir sport fishery to determine whether some fish enter the reservoir as age-0 fry. Collection of additional water samples and juvenile bull trout may be required in 2007 to confirm microchemistry signatures for individual ALR tributaries.

Key Words: rainbow trout, bull trout, snorkel surveys, abundance estimates, density estimates, redd counts, monitoring program

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1.0 INTRODUCTION

Bull trout (*Salvelinus confluentus*) are the dominant large piscivorous salmonid in the Arrow Lakes Reservoir (ALR), with a recent estimated population of 7,500 catchable-sized fish (Sebastian et al. 2000). Bull trout support an annual harvest averaging about 1,000 individuals (Arndt 2004). The molecular genetic analysis of Latham (2002) suggested at least two genetic units for conservation and management. Bull trout populations north of MacDonald Creek tend to have 'northern' genotypes that probably reflect being founded from the Upper Kootenay drainage, whereas MacDonald and other 'southern' tributaries have bull trout genotypes more representative of a southern or western founding source that colonized via the Columbia mainstem.

The original Arrow Lakes were impounded by the construction of the Hugh Keenleyside Dam at Castlegar in 1969, which raised the mean water level by 12.5 m (Pieters et al. 1998). Adfluvial bull trout losses, from construction of this dam and others (Mica Dam built in 1973 and Revelstoke Dam built in 1984) along the Canadian portion of the Columbia River mainstem, probably occurred, but estimates of dam impacts were made using only limited information concerning the distribution and production of adfluvial stocks in tributary habitat (Sebastian et al. 2000). Major impacts have included flooding of lower tributary reaches, decreased nutrient input from upstream sources, and, most significantly, fragmentation of mainstem habitat including elimination of fish migration from the Arrow Lakes to spawning tributaries upstream of Revelstoke Dam.

Prior to the construction of the Revelstoke Dam, biologists conducted cursory assessments of available bull trout rearing habitat upstream of the dam site, and concluded that insufficient habitat existed to replace losses that would be caused by dam construction (reviewed in Sebastian et al. 2000). A bull trout hatchery program for the ALR was therefore initiated in the 1980s to mitigate assumed reductions in natural production. Recent studies (Latham 2002, Arndt 2004) have suggested that hatchery enhancement has contributed little to overall production of bull trout in the reservoir.

Natural production appears to be supporting stable catch [1]rates for bull trout in the ALR sport fishery (Arndt 2004). The hatchery program was discontinued after 2000. Little is known about the population dynamics of adfluvial bull trout in the ALR or the relative contribution of the various spawning tributaries to juvenile recruitment in the reservoir. In many cases, the more basic question of which tributaries actually support adfluvial bull trout populations remains unanswered.

The Columbia Basin Fish and Wildlife Compensation Program (CBFWCP) and BC Hydro Columbia Basin Generation wish to develop a better understanding of natural production of bull trout in the reservoir, so that the success of nutrient enrichment and other compensation activities can be evaluated, and possible conservation threats, limiting factors, and enhancement opportunities identified. There are a number of inherent challenges in examining bull trout production from tributaries in the ALR. Some tributaries known to be used by adfluvial bull trout, such as the Illecillewaet and Incomappleaux Rivers, pose serious difficulties for monitoring juvenile abundance or adult escapement because of turbid water and the considerable size of mainstem reaches. In many other tributaries, the number of accessible sampling locations is very limited, especially in reaches downstream of migration barriers used by the adfluvial populations from the reservoir.

To be useful, any monitoring program conducted in ALR tributaries must meet two important criteria. First, the subset of tributaries (or tributary reaches) chosen for monitoring must contribute substantially to overall bull trout population in the ALR. If the subset does not include at least some tributaries from the core of the range, the population trends and habitat associations observed may not be representative of production dynamics for the reservoir as a whole and may lead to misleading conclusions. For example, Elliott (1987) suggested that density independent mortality factors, such as the timing and magnitude of floods, regulate populations at the margins of a species range, while density dependent factors regulate populations in core habitats.

[1] Bull trout populations in Revelstoke and Kinbasket reservoirs also provide productive sport fisheries (K. Bray, BC Hydro, Revelstoke, pers. comm.).

Second, for abundance monitoring to be capable of rapidly detecting changes in the population state, relatively precise indices of abundance are required (Korman and Higgins 1997; Ham and Pearsons 2000). This is true for both juvenile and adult life stages. Therefore, reliable assessment methods are needed as well as adequate sampling effort in each monitored tributary or reach to ensure reasonably precise estimates. Annual indices of juvenile and/or adult abundance, providing these indices were sensitive to population change, could serve as performance measures for assessing the response of bull trout to reservoir fertilization and other compensation initiatives, and also help identify limiting factors to bull trout production in the ALR.

For the purposes of a pilot study in 2004 (Decker et al. 2005), CBFWCP and BC Hydro biologists identified five tributary drainages to be evaluated as candidates for juvenile and adult spawner monitoring programs. These were selected from the larger number of tributaries for which adfluvial bull trout use was known or suspected based on previous biological studies (e.g. Bray and Mylechreest 1999; Sebastian et al. 2000), anecdotal information, or the professional judgement of CBFWCP and BC Hydro staff. Tributaries were selected to provide representation of the Upper and Lower Arrow Lakes portions of the reservoir, the northern and southern bull trout haplotypes, and small to large streams. Tributaries surveyed in 2004 included Taite Creek, and the Burton/Caribou/Snow complex, which drain into the lower basin, and MacDonald (Slewiskin) Creek and the Halfway and Jordan Rivers, which drain into the upper basin (Figure 1a). With the exception of the Jordan River, all of these streams rise in the Selkirk Mountains on the east side of the reservoir (Jordan River drains the Monashee Mountains to the west). During the 2004 pilot study, the list of tributaries was expanded to include Kuskanax Creek (Figure 1a), a non-glacial stream that was thought to support an adfluvial bull trout population (McPhail and Murray 1979), and Greeley Creek, a tributary of the Illecillewaet River where bull trout was known to occur (S. Latham, pers. comm.).

In 2004, mark-recapture methodology was used to evaluate snorkeling counts as a population assessment tool for juvenile bull trout, and redd counts were evaluated as a monitoring tool to assess adult escapement (Decker et al. 2005). Results from 2004 indicated that the distribution of bull trout in the streams surveyed was considerably less extensive than had been assumed prior to the study. Among the streams sampled in 2004, we identified only four core adfluvial bull trout reaches (i.e., reaches where juvenile densities were comparable to densities reported for known adfluvial bull trout systems): upper Halfway River, and upper Caribou, upper McDonald and Greeley creeks.

The first objective for the 2005 and 2006 study years was to complete the evaluation of whether juvenile snorkel surveys and adult redd surveys could provide reliable estimates of bull trout abundance. The other major objective was to complete the process of identifying tributary reaches that support adfluvial bull trout populations, and to assess the relative importance of each to total production in the ALR. Results from 2004 pilot study indicated that biophysical conditions and/or interspecific competition may be important limiting factors for bull trout production in the ALR, and suggested that glacial systems (Incomappleaux and Illecillewaet Rivers), which were not sampled due to poor water clarity during late summer when other tributaries were sampled, may be important contributors to overall production (Decker et al. 2005). Because major information gaps also exist with respect to natural production of rainbow trout in the ALR - which are also important to the sport fishery – as much as possible, we collected abundance and snorkeling accuracy data for this species during juvenile snorkeling surveys in ALR tributaries as well. During 2005 and 2006, we excluded reaches sampled in 2004 that were used only marginally by adfluvial bull trout and/or possessed biophysical conditions that limited their potential use (Taite, Burton, Snow, Jordan), and concentrated on new reaches in the glacial Illecillewaet and Incomappleaux river systems. This report details new findings in 2006 and summarizes the results of the study to date (2004-2006) for adfluvial bull trout and rainbow trout.

2.0 METHODS

2.1 Study area

Latham (2002) provides a review of the geology and ecology of the ALR and of recent anthropogenic disturbances in the catchment. He also details the genetic history and diversity of bull trout in reservoir. Briefly, the ALR is situated between the Monashee and Selkirk Mountains in the southern interior of British Columbia and is located on the mainstem of the Canadian portion of the Columbia River. The reservoir and its tributaries lie mainly within the Interior Cedar-Hemlock biogeoclimatic zone (Krajina 1959), with upper tributary reaches extending into the Engelmann Spruce-Subalpine Fir and Alpine Tundra zones. Summers are typically cool with moderate rainfall, while winters are cold with substantial snowfall. Tributary hydrographs are snowmelt driven, with peak flows during the June freshet followed by low summer and winter flows. Summer flows are elevated by modest runoff from permanent snowfields in many tributaries, and by considerable runoff from glaciers in several larger, northern tributaries (Illecillewaet, Incomappleaux, Jordan).

The landscape adjacent to the ALR is steep and rugged. Glacial erosion has resulted in hanging valleys evidenced by waterfalls that are present in most tributaries (Latham 2002). Southern tributaries (Kuskanax, MacDonald, Caribou, Burton, Snow, Taite) tend to be short and have waterfall barriers near their mouths. Relatively high average gradient in these tributaries is reflected in highly confined channels with step-riffle-pool morphology, and predominately riffle and cascade habitat. Larger tributaries draining into what was historically Upper Arrow Lake (Halfway, Incomappleaux, Illecillewaet) are generally lower in gradient, have alluvial channels with riffle-pool morphology, and have waterfall barriers further from the reservoir.

In the Illecillewaet and Halfway rivers, migration barriers (one human-made and one natural) were altered to allow fish from the reservoir access to upstream habitat. This was done in an attempt to compensate for tributary habitat flooded by dam construction and the raising of the Arrow Lakes. In the Illecillewaet River, fish migration was blocked by the construction of a hydroelectric dam on that river in 1898, in a canyon located 2.3 km from the mouth (McBurney and Udell 1977). Prior to this, adfluvial bull trout from the reservoir likely had access to about 39 km of mainstem habitat upstream of the canyon, but from 1898 until dam removal in 1977 bull trout persisted in the upper river in a resident fluvial form only (Northern Natural Resources Services 1976). Stocking of adfluvial bull trout in Illecillewaet River and other tributaries occurred sporadically from the 1970's until the termination of the bull trout hatchery program after 2000. In the Halfway River, a natural barrier located 9.9 km upstream of the reservoir was altered with explosives in the 1980's. This allowed adfluvial bull trout to access the upper river, which had historically supported the resident life-history form only (Latham 2002). Our results suggest that other migratory species from the reservoir, including rainbow trout (*Oncorhynchus mykiss*), kokanee (*O. nerka*), and mountain whitefish (*Prosopium williamsoni*), are unable to ascend what are now major obstructions at the sites of the former barriers in these streams (resident mountain whitefish are present in the Illecillewaet above Box Canyon).

Fish assemblages below barriers may include adfluvial bull trout, rainbow trout (resident or adfluvial), kokanee, mountain whitefish, longnose dace (*Rhinichthys cataractae*) and sculpin (*Cottid* species) (Latham 2002). Above barriers in southern tributaries, fish communities are composed almost exclusively of introduced rainbow trout and eastern brook trout (*Salvelinus fontinalis*). In northern tributaries, fish communities below barriers are similar, but above barriers, resident bull trout and slimy sculpin (*Cottus cognatus*) are more likely to be present than introduced species (Latham 2002). Native westslope cutthroat trout are present upstream of barriers in the Jordan and Akolkolex River systems.

Taite, MacDonald, Burton, Caribou and Snow Creeks are all located on the east shore of the reservoir south of the town of Nakusp (Figure 1a). Kuskanax Creek is located on the east shore at Nakusp, and Halfway River is located on the east shore to the north of Nakusp. The Jordan River is located on the west shore immediately downstream of the Revelstoke Dam. The Illecillewaet River enters the reservoir near Revelstoke, and has Greeley, Albert, West Twin, and Woolsey creeks, and Tangier River as major tributaries below its barrier (Figures 1b-e). The Incomappleaux River enters the reservoir on its east shore at the head of Beaton Arm near the Shelter Bay Ferry Terminal (Figure 1a), and has numerous tributaries below its barrier (Pool, Menhinick, Sable, Boyd, Kelly, McDougal, Battle Brook, Lexington; Figures 1f-i). Barriers to fish migration in ALR tributaries were not well defined in previous studies. As a starting point, we searched available reports and stream inventory databases for the geographic locations of potential barriers and determined whether the absence of adfluvial bull trout upstream of these locations had been confirmed. During the study, we attempted to locate these barriers and other potential barriers downstream. In cases where we observed adfluvial bull trout above reported barriers, we searched upstream for the actual barrier. For each tributary, stream length below the barrier was estimated using BC Terrain Resource Information Management (TRIM) data.

2.2 Juvenile survey

2.2.1 Juvenile sampling design

In the original study design, individual tributaries were to be treated as whole units (i.e., no reach stratification). However, as juvenile surveys progressed during 2004-2006, abrupt changes in fish abundance and age structure were noted for bull trout and rainbow trout above and below major obstructions (passable by adfluvial adult bull trout and located downstream of the actual migration barrier) in Caribou Creek (2.5-3 m falls), Halfway River (a series of three 2-2.5 m falls; Appendix 9, plate 1), Illecillewaet and Incomappleaux rivers (major falls; Appendix 9, plate 3), and in upstream and downstream sections of MacDonald Creek (Decker et al. 2005). To address this we delineated upstream and downstream reaches in these tributaries and increased sampling effort so that at least two sites were sampled in each reach. The remaining tributaries (Kuskanax, Taite, Burton, Jordan, Greeley) were treated as single reaches. Throughout this report we use the terms tributary and reach interchangeably when referring to the study reaches.

In 2005 juvenile surveys were not conducted in Kuskanax, Taite, and Burton Creeks and the Jordan River because sampling in 2004 indicated only marginal use by juvenile bull trout. Surveys were repeated in Caribou, MacDonald, and Greeley creeks, and Halfway River in 2005, and new sampling was carried out in Illecillewaet River in November when the river had cleared. None of the juvenile sites from 2004 and 2005 were resampled in 2006, but new sites were sampled in Illecillewaet and Incomappleaux rivers and their tributaries during clear water conditions in late October - early November.

Throughout the study area, the smaller reaches that were sampled were relatively steep (> 2%), and had a channel morphology consisting of a series of "stepped" riffles and cascades interspersed with short pools and shallow runs. With the exception of the Illecillewaet and Incomappleaux mainstems, larger reaches consisted of small-to-medium sized 'pockets' of juvenile fish habitat along the margins of a relatively swift and steep mainstream channel. Therefore, despite other studies indicating that pools are preferred habitat for juvenile bull trout (McPhail and Murray 1979; Saffel and Scarnecchia 1995; Hagen 2000), in our case, stratification of sampling by riffle and pool habitat was inappropriate. Alternatively, we sampled relatively large sites (> 50 m in length) that included habitat sequences in the smaller reaches and series of 'pocket habitat' in the larger reaches. In addition, because fish densities were low, particularly for bull trout, larger sites allowed for sufficient numbers of fish to be captured and marked in order to estimate snorkeling efficiency (proportion of marked fish seen). In contrast, the much larger, lower gradient Illecillewaet and Incomappleaux rivers have a riffle-pool (or riffle-run) channel morphology, and we delineated three habitat strata to be sampled at each survey location: riffles, runs and pools (combined as one strata), and small braids and side channels (combined as one strata).

We limited the juvenile component of the study to bull trout and rainbow trout that were age-1+ and older (i.e., parr and adult residents). Other studies have shown that snorkeler counts are not effective for estimating the abundance of young-of-the-year salmonids (Griffith 1981; Campbell and Neuner 1985; Heggenes et al. 1990; Hillman et al. 1992). Sampling bull trout fry abundance is notoriously difficult, regardless of the method used because of their benthic orientation (Baxter and McPhail 1996; Peterson et al. 2004). For bull trout, parr abundance is likely to be a better index of stream carrying capacity than fry abundance because migratory salmonids that have a lengthy stream residence typically experience the majority of density-dependent mortality in their first year of life (Elliot 1987; Kennedy and Crozier 1993; Ward and Slaney 1993). In the

Thutade watershed in northcentral British Columbia, bull trout fry (age 0+) abundance exhibits substantial interannual variability, while parr abundance is relatively consistent from year-to-year in this unexploited system (Bustard 2004).

2.2.2 Juvenile sampling methods

2.2.2.1 Snorkeling surveys

Electrofishing and underwater observation by snorkelers are the two most common methods for sampling bull trout parr (Thurow and Schill 1996), but both have disadvantages with regard to ALR tributaries used by adfluvial populations. Conventional multiple-pass electrofishing is very time consuming and resources are often insufficient to allow for sampling adequate to ensure acceptable levels of precision. Research has demonstrated that spatial variation in fish density is often high relative to the measurement error associated with estimating fish abundance (Hankin and Reeves 1988; Decker et al. 1999). Thus, sampling larger sites or a greater number of sites using a more rapid but less precise method (such as snorkeling surveys) can often provide more precise abundance estimates for a given cost. A second major disadvantage of electrofishing is that it is frequently impossible to achieve good results for older juvenile age classes in all but the smallest streams because mainstem habitats cannot be enclosed with stop nets or electrofished effectively. None of the ALR tributaries included in our study were small enough to be spanned with stop nets during 2004, thus, sampling three-sided margin sites, in which parr have a greater chance of being disturbed and displaced from the site, was the only option for electrofishing. Furthermore, multi-pass electrofishing has been shown to underestimate fish abundance because of declining capture probabilities for subsequent passes (Riley et al. 1993; Peterson et al. 2004). This problem may be particularly acute for bull trout parr, which are benthic-oriented and are often found in streams with large substrate and very low conductivity. Peterson et al. (2004) found that removal estimates can under-estimate true abundance of bull trout parr by as much as 88%.

As an alternative to electrofishing, we used calibrated nighttime snorkeling counts to estimate parr abundance. The use of snorkeler surveys can increase sampling efficiency several-fold compared to conventional multi-pass electrofishing (Roni and Fayram 2000). Snorkeler counts can provide reliable estimates of fish abundance if calibrated using more accurate methods such as mark-recapture at a portion of the sampling sites (Hankin and Reeves 1988; Thurow and Schill 1996; Hagen et al. in prep.). In this study we employed snorkeling counts to estimate parr abundance in all study tributaries.

Daytime concealment behaviour is common in juvenile salmonids (e.g., Bradford and Higgins 2000 and references therein), and is likely an important factor affecting the reliability of snorkeling surveys. Daytime concealment behaviour likely depends on factors such as temperature, time-of-day, season and habitat. Variability in daytime concealment behaviour may partly explain differences in the accuracy of daytime snorkeler counts among streams (Cunjak et al. 1988), or within streams at different times of the day (Thurow and Schill 1996) or at different temperatures (Hillman et al. 1992). We therefore conducted our snorkeling surveys at night, limiting surveys to a four-hour period beginning 0.5 hours after dusk. Bradford and Higgins (2000) found that in the Bridge River, throughout the year, the highest abundances of steelhead parr were always observed during the 4-hour period following dusk.

To illuminate the sampling sites at night, snorkelers used handheld dive lights. In clear, small streams snorkelers attempted to reduce the disturbance to fish by reflecting the light beam off the surface or by using sanded plastic filters over the lens. Snorkelers worked in groups of two, with each snorkeler entering the site at its downstream end and systematically sweeping in an upstream direction the area between their bank and the agreed upon mid-point of the site. In the larger study reaches it was often not necessary for snorkers to search the entire width of the stream channel because midstream velocities were too great to provide juvenile fish habitat at night (see Edmundson et al. 1968; Campbell and Neuner 1985; Hagen and Taylor 2001; Muhlfeld et al. 2003). However, areas where mid-channel boulders or debris provided velocity breaks were searched thoroughly.

The fork lengths of all juvenile bull trout and rainbow trout observed that were large enough to potentially be age-1 parr (> 60 mm) were visually estimated and recorded in waterproof notebooks, while juveniles smaller than this were tallied as fry, without estimates of their lengths. To aid in the estimation of fish length, snorkelers drew ruled scales on the cover of their notebooks. At night, snorkelers were typically able to hold the notebooks within 30 cm of a fish to measure its length without disturbing it. Mountain whitefish were tallied by size class (juveniles: < 200 mm; adults or sub-adults: > 200 mm), and all other salmonids were recorded as being present or absent.

2.2.2.2 Calibration of snorkeling surveys

During the 2004-2005 mark-recapture study we investigated the among-site variation in snorkeling efficiency for bull trout and rainbow trout (q: proportion of marks seen by snorkelers) at a portion of the juvenile sampling sites in order to evaluate how reliably snorkeling efficiency estimates could be extrapolated to other sites and reaches in our study area. Effort was made to distribute the calibration sites among the study reaches. At each calibration site, fish were captured for marking using one of two methods. In the first, a snorkeler captured bull trout and rainbow trout parr one night prior to the snorkeling survey described above. We chose to conduct the underwater surveys roughly one day after marking because we considered this to be the shortest time period that would still allow fish to recover from marking and complete a diurnal cycle of movement and redistribution within the site, but would minimize movement from the site. Fish were captured using two large aquarium nets affixed to wooden handles of approximately 80 cm in length. Our methods assume that fish throughout the site had an equal probability of receiving a mark (Ricker 1975). To ensure that this assumption was not greatly violated, during marking fish were netted from all areas in which they were encountered, with no areas omitted from the search irrespective of the effort required. The snorkeler worked slowly through the site, locating and capturing fish. In near-shore areas too shallow to access from an underwater position, parr were searched for and captured by walking slowly through the habitat. Captured fish were handed to a second crew member on shore, who immediately measured the fish's fork length, removed scales if required for aging analysis (from a location approximately 2-4 rows above the lateral line and between the back of the dorsal fin and the origin of the anal fin), and marked it.

In the second method fish were captured during daylight hours, 28-32 hours prior to the snorkeler survey, by selectively electrofishing the site at potential parr holding locations. This differed from typical removal electrofishing in that extensive application of electricity to the site was avoided in an attempt to minimize disturbance to fish prior to the snorkeling survey. Fish captured by electrofishing were allowed to recover in the sampling bucket and then placed in near-shore, zero-velocity areas adjacent to their original lies.

For both capture methods, marking consisted of inserting a size 16, barbed fish hook attached to a 15 mm long piece of coloured plastic chenille through the skin of the fish's back at the anterior or posterior insertion of the dorsal fin (Appendix 9, plate 2). In 2005 marking differed from that in 2004 in that size-specific marks were deployed. For rainbow trout, three size categories were marked: <100 mm, 100-170 mm, and >170 mm, whereas for bull trout, two marking categories were used: <110 mm and >110 mm. Captured fish were not anaesthetized because of uncertainty about behavioural effects from the anaesthetic, and because they could be measured and marked without it. Immediately following marking, fish captured by snorkelers were returned to the original lie they had been captured in, although not before the snorkeler had moved beyond it.

During snorkeling surveys the following evening, marked and unmarked parr were recorded separately, so that snorkeling efficiency (q: percentage of marked fish seen) could be estimated. This type of mark-recapture study assumes a closed population, whereas our sites were not enclosed. Over sufficiently short time periods, however, and if study animals restrict their movements to a defined area, physically open sites can be treated as closed without introducing significant bias (Pollock 1982; Mitro and Zale 2002). Bias can also be minimized by increasing the length of the recapture site relative to the marking site (Albanese et al. 2003). We investigated the assumption of site closure by surveying an additional distance of approximately half the site length adjoining both the upstream and downstream site boundaries, so that the total distance surveyed for marks was approximately two times the marking site length. Marked fish that had moved beyond the original site boundaries were recorded separately. We disregarded unmarked fish observed in the upstream and downstream sections adjacent to the original marking site, assuming balanced immigration and emigration of unmarked fish from the marking site (Albanese et al. 2003).

To reduce the unwanted effects of binomial variation associated with too small a number of marks, we had as a target 15-20 marks per site, with 10 marks considered a minimum. This represents a trade-off between achieving relatively precise estimates of mean site-level snorkeling efficiency (i.e. maximum number of sites) versus precise estimates of snorkeling efficiency at individual sites (maximum number of marks per site). We combined mark-recapture data from sites with fewer than 10 marked parr with mark-recapture data from the closest mark-recapture site in the same reach to give a single estimate of site-level snorkeling efficiency for the two sites.

2.2.2.3 Habitat characteristics at sampling sites

Habitat measurements at sampling sites were made during the daytime prior to conducting the snorkel surveys at night. Physical site attributes recorded during site layout included 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 area covered by wood debris greater than 10 cm in diameter or 1 m^2 in surface area), channel confinement, substrate composition (boulder, cobble, gravel, and fines as percentages of the site area), D90, D50, site length, wetted width, channel width, water surface gradient (measured using an automatic level and a surveyor's rod) and thalweg depth (average of 10 measurements, each taken at the deepest point along a cross-channel transect). D90 and D50 refer to, respectively, the estimated diameters of substrate particles for which 10% and 50% of the site area consists of larger particles.

2.3 Adult bull trout survey

2.3.1 Study design

As an index of adult adfluvial bull trout escapement we employed the widely used method of visual counts of redds, or excavations in the substrate associated with spawning activity and egg deposition (Rieman and McIntyre 1996; Rieman and Myers 1997; Dunham et al. 2001). Redd counts are one of the least expensive and least invasive of adult population assessment methods, and can be reliable indicators of abundance (Dunham et al. 2001), yet there are a number of considerations for their application. Rieman and Myers (1997) reviewed times series of redd count data from Idaho and Montana and found that variation in redd counts among observers/surveys made the detection of trends in individual streams unlikely over limited time scales. Dunham et al. (2001) found that although redd count data were strongly correlated with actual abundance, inter-observer variability was nonetheless a significant source of error in redd count accuracy and precision, although it should be noted that Muhlfeld et al. (2006) documented substantially lower levels of observer variability when experienced observers were used.

As part of a long-term bull trout monitoring program in Thutade Lake watershed in northcentral British Columbia, Bustard (2004) made a substantive effort to reduce the inter-observer variability in redd counts by establishing detailed criteria for redd identification and providing training to new crew members at the beginning of each annual survey. We followed Bustard's criteria for discriminating redds (see below) and used experienced field crew to conduct the surveys. Lacking prior information about the distribution of spawning in the study tributaries, redd surveys proceeded downstream from migration barriers to the stream mouth, or until the redd encounter rate had diminished to negligible levels.

2.3.2 Redd survey methods

Redd surveys were conducted by two observers wearing waders and polarized sunglasses. Observers walked downstream parallel to one another on either side of the stream, or offshore in order to gain the best view of potential spawning locations. In highly confined canyon reaches (lower Halfway, Kuskanax, Caribou, Jordan), one observer wore a drysuit, mask and snorkel, so that he could investigate deeper or more turbulent areas.

Redds were identified as approximately dish-shaped excavations in the bed material, often of brighter appearance than surrounding substrates, accompanied by a deposit beginning in the excavated pit and spilling out of it in a downstream direction. Disturbances in the bed material caused by fish were discriminated from natural scour by: i) the presence of tail stroke marks; ii) an over-steepened (as opposed to smooth) pit wall often accompanied by perched substrate that could be easily dislodged down into the pit, and often demarcated by sand deposited in the velocity break caused by the front wall; iii) excavation marks alongside the front portion of the deposit demarcating the pit (bull trout can deposit eggs in more than one event as the redd is built in an upstream direction; Leggett 1980); and iv) a highly characteristic overall shape that included a 'backstop' of gravel deposited onto the unexcavated substrates, a deposit made up of gravels continuous with this backstop and continuing upstream into the pit, and a pit typically broader than the deposit and of a circular shape resulting from the sweeping of gravels from all sides to cover the eggs (in a portion of redds gravels are swept into the pit from only one side, often a shallow gravel bar on the shore side).

A second important determination was whether fish had actually spawned at a location where an excavation had been started. 'Test digs' were considered to be pits, often small, accompanied by substrate mounded up on the unexcavated bed material downstream but with no substrate swept into the pit itself, which would denote at least one egg deposition event. Redds can be small, as female bull trout can spawn in more than one redd if the substrate conditions at the first location are not optimal (Leggett 1980). In the case of a 'test dig' determination the mound of gravels would typically be short and narrow around the downstream side of a relatively small pit. In the Thutade watershed gravel deposits associated with test digs of this description have been excavated and few have been found to contain eggs (J. Hagen and D. Bustard, personal observation).

In areas of limited gravel or high redd abundance, or where spawning site selection is highly specific, superimposition of redds upon one another can occur (Baxter and McPhail 1996). When superimposed redds were encountered, we based our counts on a subjective evaluation, with the most recent complete redd(s) counted and the disturbed remains of prior redds being estimated in relation to it. A greatly extended deposit length (subjectively evaluated to be at least twice the length of a 'typical' deposit length) was grounds to consider whether other females had made use of the pit created by a first to construct additional redds. Block (1955, as cited in Fraley and Shepard 1989) observed one male bull trout spawn with three females in succession at a single redd location, which expanded each time. Fortunately, such cases usually represent a small proportion of the total number of redds present. All redd locations were recorded using hand-held GPS units. In 2004 and 2005, the length and width (total width of disturbed area measured across the top of the deposit) of each redd were also estimated.

2.4 Water sampling

In 2006 we collected replicate (2) water samples at two locations in the reservoir, and at one location in Caribou, MacDonald, Halfway, Incomappleaux, and Illecillewaet tributaries. Water samples were collected to assess the concentration of several elements, including strontium, barium, zinc, manganese, magnesium, and calcium. Fish residing in a stream will have ratios of strontium, barium, zinc, manganese, and magnesium to calcium in their otoliths that are similar to ambient ratios in the water where they reside (Bacon et al. 2004). If there is sufficient contrast in the ratio of any of these elements to calcium between juvenile rearing tributaries and the reservoir, microchemistry and laser ablation techniques can be used to analyze adult otoliths and determine age-at-reservoir-entry. This analysis may be warranted in the future to examine whether some bull trout enter the reservoir at age 0+ and successfully contribute to the spawning population. This has implications with respect to whether tributary rearing space represents a bottleneck to adult abundance in the reservoir (McPhail and Murray 1979). CBFWCP has already accumulated a collection of adult bull trout otoliths from the sport fishery in the reservoir. Results of water sampling are not discussed in this report, but the data are included in Appendix 8).

2.5 Data analyses

2.5.1 Juvenile length-at-age

To estimate discrete length categories for each age class of bull trout and rainbow trout, we relied on both frequency histograms of fish fork length and scale samples. Fish captured for marking by snorkelers and electrofishing crews were measured and sampled for scales, but most length data, including all data collected during the actual abundance surveys, was derived from visual estimates made by snorkelers. Therefore it was important to assess whether snorkeler estimates of fish length were accurate enough to generate age class-specific estimates of abundance. To test this we cross-validated the two methods of estimating length categories (frequency histograms and scale aging) by having two different individuals conduct the analysis for each method independent of one another, and then comparing their estimates of length range for each age class. Comparable (i.e., within 10 mm) estimates of length range for each age class for the two methods and the two individuals would suggest that snorkelers could estimate the length of observed fish well enough that fish ages could be reliably determined.

Collected scales that were suitable for analysis were identified under 47X magnification on a microfiche reader-printer, and photographs were taken. Regions of closely spaced circuli on the scale were identified as annuli. Each photographed scale was measured along the focus-anterior axis, the radius of each annulus and the outer scale margin being recorded. In order to improve our understanding of growth and potential minimum and maximum length for a particular age class, we also investigated the relationship between fish length and scale radius for bull trout and rainbow trout using simple linear regression (Zar 1996). Lengths-at-age were then back-calculated for each scale according to the Fraser-Lee equation (Duncan 1980):

$$l_k = c + (L - c) r_k / R \tag{1}$$

where:

l_k is the length at age k

c is the constant of proportionality from the fish length/scale diameter regression

L is the fish length at time of capture

r_k is the radius of the annulus at age k

R is the scale radius at the time of capture

2.5.2 Mark-recapture estimates of snorkeling efficiency for juveniles

Estimates of snorkeling efficiency are proportions, and are binomially as opposed to normally distributed, with the deviations from normality being substantial for small or large proportions (0 to 0.3 and 0.7 to 1.0; Zar 1996). Resultant data (q_i'), however, will have a nearly normal distribution if the original proportion observed at each site (q_i) is transformed using the arcsine square root procedure, where:

$$q_i' = \arcsin\sqrt{q_i} \quad (2)$$

To assess whether normal distribution statistics could be used to model untransformed or arcsine square root-transformed site-level snorkeling efficiency data, we computed Kolmogorov-Smirnov one-sample tests of normality (Zar 1996). Separate estimates of mean snorkeling efficiency and 95% confidence intervals were computed for the two species (bull trout and rainbow trout) and two types of mark-recapture data collected in the ALR tributaries (marked populations captured by snorkeling and marked populations captured by electrofishing).

We first examined whether data from the two years could be combined to improve sample size, and whether the two species should be treated separately, using two-sample *t*-tests (Zar 1996) for species- and year-specific estimates of snorkeling efficiency, with site as the sample unit.

For each species we investigated whether snorkeling efficiency was related to capture method, body size, and habitat variables measured at mark-recapture sites using logistic regression with data pooled from all mark-recapture sites. First, a correlation matrix for the suite of habitat variables recorded at sampling sites was constructed to investigate the potential for autocolinearity. Variables were eliminated from the analysis if bivariate correlations with other variables describing similar attributes (e.g., D90 versus substrate composition) were greater than 0.7 (Tabachnick and Fidell 2001). Direct logistic regression (all variables enter the analysis simultaneously) was then conducted to evaluate the effects of the resulting set of habitat variables (plus dummy-coded variables representing categorical body size in 2005 and capture method in both years) on the likelihood of marked fish being seen.

Because of evidence of autocolinearity among predictor variables, the significance of individual predictors was evaluated using stepwise logistical regression and a backward elimination procedure where the predictor with the highest *P* was eliminated from the analysis and the regression re-calculated, continuing until the *P*s for all remaining predictors in the regression were statistically significant (Zar 1996). However, stepwise logistic regression was conducted only if the direct logistical regression with all the predictors had been statistically significant.

Chi-square tests were used to test whether the rate of movement of marked fish from the original marking site into the adjacent upstream and downstream sections was dependent on which method of capture was used during fish marking (snorkeling or electrofishing), and to test whether the length distributions of snorkeler- and electrofishing-captured marked populations differed.

2.5.3 Bull trout and rainbow trout parr abundance estimates

To provide calibrated estimates of bull trout and rainbow trout abundance at each site (including sites without mark-recapture data), we divided the snorkeling count for each species by estimated snorkelling efficiency. For each species, mean snorkeling efficiency (q_{mean}) was estimated by averaging snorkeling efficiency estimates at individual mark-recapture sites (q_i). In cases were snorkeling efficiency varied significantly among fork length categories, mean snorkeling efficiency estimates were size category-specific. Confidence intervals for fish abundance at individual sites were estimated by expanding the snorkeler counts using the upper and lower 95% confidence limits for snorkeling efficiency in place of *q*:

$$\pm 95\%\ CI(q) = q \pm 2\sqrt{Var(q)} \tag{3}$$

If the objective is to estimate fish abundance for the entire study area[1], as opposed to abundance at individual sites, the confidence interval would be computed using the standard error of the estimate instead of the standard deviation:

$$\pm 95\%\ CI(q) = q \pm 2\sqrt{\frac{Var(q)}{n}} \tag{4}$$

This properly reflects the number of mark-recapture sites distributed among the tributaries that contributed to the overall estimate of mean snorkelling efficiency.

Fish abundance per unit area is reported as both linear density (parr/100 m) and density by area (parr/100m^2). Density by area is more commonly reported in the literature and facilitates comparison of ALR results with other studies, but linear density provides a more meaningful comparison of parr abundance among ALR tributaries because the larger streams, despite having much greater wetted area for a given length, do not necessarily provide more usable area since juveniles are confined to stream margins by high mid-stream velocities (Decker et al. 2005).

For each species and age class, mean fish density (fish/100m^2 and fish/100 linear m) for a particular reach was computed as the mean for all sites in that reach. The number of sample sites in individual reaches was too low (2-5 sites; see section 3.2) to assess spatial variation in fish density. Therefore, we did not compute 95% confidence intervals for

[1] An objective of this study was to determine whether snorkelling counts could provide a reliable index of year-to-year abundance of juvenile bull trout in ALR tributaries. Therefore, 95% confidence intervals for snorkelling efficiency applied to the entire study area (Equation 4) are included to give an indication of the amount of uncertainty that snorkeling counts would contribute to a single index value representing total juvenile standing stock for the system.

reach-level fish density estimates; they are presented as point estimates for the sake of discussion, but the reader should be aware that these estimates may not accurately reflect true abundance. To compare relative juvenile production among the study reaches, we also computed estimates of bull trout and rainbow trout parr standing stocks for each reach by simply multiplying linear fish density by reach length. Again, these estimates are only approximations, but they allow for a comparison of the relative contribution of individual reaches to juvenile production in the ALR as a whole.

3.0 RESULTS

3.1 Reconnaissance of migration barriers

Ground reconnaissance in August 2004 confirmed previously reported locations for migration barriers in MacDonald, Caribou, Taite and Kuskanax creeks and Jordan River (Figure 1; see Appendix 7 for geographic coordinates and accessible stream lengths). Due to their location in deep bedrock canyons, we were unable to access possible barriers in Burton and Snow creeks that were identified during a previous helicopter reconnaissance (James Baxter, BC Hydro, data on file), although the coordinates for the barrier in Burton Creek (49° 55' 42.4" N, 117° 14' 36.2” W) were clearly incorrect, as they indicated a position on the opposite side of the reservoir. Given the continuous, high gradient nature of these streams within and above the canyons (based on 1:50000 topographic maps), fish migration from the reservoir is most likely limited to the lower few kilometres of these streams.

In Caribou Creek, an additional 3 m high waterfall was located 4.1 km upstream of the reservoir and 1.8 km downstream of the barrier (see Appendix 7 for coordinates). We observed adfluvial adult bull trout upstream of this obstruction during the August 2004 juvenile survey, but not during the August 2005 survey (high flows prevented redd surveys in 2004 and 2005). In July 2005 forestry personnel observed 30-40 dead adfluvial adult bull trout that had unsuccessfully attempted the falls at 4.1 km and then fallen into a small isolated pool where they succumbed to oxygen deprivation (J. Burrows, Ministry of Environment, pers. comm.). Why adults encountered difficulty in ascending these falls in 2005 is unclear, but low flows or a change in the morphology of the falls may have been contributing factors. Prior to the 2006 spawning migration, local resource agencies and volunteers coordinated to fill the isolated pool with concrete. Although we did not observe any dead adults at the falls during redd surveys in 2006, few spawners ascended the falls (see section 3.3.2), and these fish likely consisted of the early portion of the run. The falls appeared to be a migration barrier for adults arriving in August and September when flows were relatively low.

In Halfway River, no migration barriers were found in the lower 33 km. Adfluvial bull trout redds were observed upstream of the falls at 9.9 km, indicating that blasting in the 1980s rendered this former barrier passable, at least to some fish. In each of the two headwater forks of Halfway River that meet at 32.5 km from the stream mouth (Figure 1j), we did not locate any barriers in 500 m long sections upstream of the mainstem; further upstream reconnaissance was abandoned due to a lack of road access. However, stream gradient increased rapidly in an upstream direction in both forks, which may limit further migration of adfluvial fish (see section 3.3.1). The north fork has reduced gradient above its initial steep climb from the mainstem, but whether adult bull trout access this stream reach has not been investigated.

On September 2, 2005 we conducted a helicopter reconnaissance of the Illecillewaet and Incomappleaux river systems to identify mainstem and tributary migration barriers. We confirmed previously reported mainstem migration barriers at Albert Gorge in the Illecillewaet River (39 km from the reservoir, Figure 1b) and just upstream of Battle Brook in the Incomappleaux River (40.6 km from the reservoir, Figure 1f). With the exception of Sable Creek, all tributaries large enough to support adfluvial bull trout populations in Incomappleaux River (McDougal, Boyd, Pool, Lexington, Battle Brook, Kelly, Menhinick, Stefanie) had obvious barriers within 1 km of their confluences with the mainstem (Figures 1f-g). There are several waterfalls in the lower canyon of Sable Creek near the mainstem, but none of these could be confirmed as barriers from the air. However, a 2006 survey of upper Sable Creek (upstream of Stefanie Creek) - which is unglaciated, relatively low in gradient, and represents potential high quality spawning and rearing habitat for bull trout - found no evidence of spawning or fish presence.

Several tributaries to the Illecillewaet River large enough to support adfluvial bull trout have barriers within 500 m of the mainstem (West Twin, East Twin, Clachnacudain) and provide relatively limited, if any, spawning habitat in the accessible sections. Greeley Creek is unglaciated and has an accessible length of 1.8 km (Figure 1c), and our juvenile and redd surveys confirmed adfluvial bull trout spawning and rearing there. Earlier studies conducted by others suggest that a series of waterfalls in the lower canyon of Woolsey Creek approximately 0.9 km from the mouth represent a migration barrier. A redd survey conducted in upper Woolsey in 2006 as part of this study also suggests this (see section 4.2). The Tangier River provides limited spawning habitat downstream of waterfalls in its lower canyon, located approximately 1.2 km from its mouth. The age structure of bull trout populations in Tangier River upstream of this canyon section, as indicated by juvenile surveys conducted as part of this study in November 2005, suggests that adfluvial bull trout do not ascend the canyon. Albert Creek has 2.7 km accessible to adfluvial bull trout. Spawning and rearing in this tributary was confirmed in 2006.

3.2 Juvenile fish survey

Most tributaries sampled in 2004-2006 had only 2-5 reasonable access points each, and even some of these were marginal. Snow Creek was excluded in 2004 due to a lack of access points. Halfway River, the most accessible stream, had 10-12 reasonable access points in two reaches. The Illecillewaet River, despite having a major highway paralleling its length, had poor access in many sections where the stream flowed through bedrock canyons. No more than 10 reasonable access points existed along the Illecillewaet River mainstem. The Incomappleaux River had a comparable number of potential access points. Poor access is due to the fact that most ALR tributaries are deeply entrenched in steep valleys, and nearby roads, if present, are separated by cliffs, steep ground, or extensive riparian wetlands. Access was particularly poor for the more remote upstream reaches where juvenile bull trout were concentrated.

During August 24 - September 3, 2004, snorkeling counts were conducted at 29 sites, in 11 reaches (Appendix 1a). We allocated the majority of sampling effort to reaches that contained substantial numbers of bull trout (determined as the survey progressed). For other reaches, once it was apparent from sampling at two or three sites that bull trout were absent, or nearly so, the objective was to conduct sufficient sampling to characterize fish populations rather than provide precise estimates of fish density. Therefore, not all accessible locations in these reaches were utilized.

During August 27-September 3, 2005, we resampled sites established in 2004 in Caribou, MacDonald and Greeley creeks (Appendix 1b). These streams were identified in the 2004 study as having core juvenile bull trout rearing reaches. Additional sites were added in 2005 to better define distributions or provide additional mark-recapture data. During November 8-10, 2005, snorkeling counts were conducted at 12 and three sites in the Illecillewaet River mainstem and its largest tributary, the Tangier River, respectively (Figures 1d-e, Appendix 1b), after glacial run-off had subsided. A major obstruction (2 m falls) located 4.1 km from the Columbia River confluence delineated upstream and downstream reaches in Illecillewaet River (Figure 1e); juvenile bull trout were not detected downstream of the obstruction.

During October 28 – November 2, 2006, new surveys were conducted in glacial mainstem and tributary reaches in the Illecillewaet and Incomappleaux systems. We sampled 24 mainstem sites that were distributed along the entire accessible length of Incomappleaux River (Figures 1h-i). An additional 10 sites were distributed among Kelly, Boyd, Sable, Menhinick, Pool, and Lexington Creeks (Figure 1h-i, Appendix 1c). A major obstruction 5.4 km from the ALR confluence (Figure 1i), above which rainbow trout were not observed, delineated upstream and downstream reaches in the Incomappleaux mainstem. In the Illecillewaet system, five new mainstem sites were sampled in 2006 along with 2 tributary sites in Albert Creek (Figure 1d, Appendix 1c).

Discharge in the study tributaries during 2004 and 2005 juvenile surveys was typical of summer low-flow conditions (Table 1), although unusually high flows did occur periodically during the survey period in 2004. As an example, late summer flows in Kuskanax Creek averaged 7.0 and 4.2 m^3/s in 2004 and 2005, respectively, compared to the 1964-2005 mean of 6.9 m^3/s (WSC, station 08NE006). Average flow during the 2005 juvenile survey in Illecillewaet River (25 m^3/s) was near the historical average for November (21 m^3/s; WSC, station 08ND013). During the 2006 surveys in the Incomappleaux and Illecillewaet River systems, discharge levels were considerably lower than the historical average (discharge in Illecillewaet River ranged from 12 to 16 m^3/s compared to a historical average of 25 m^3/s for early November (WSC, station 08ND013).

Average stream temperatures recorded during the nighttime snorkeler surveys ranged from 8°C to 13°C during the late August-early September surveys in 2004 and 2005 (Appendices 4a-b), and from 0.5 to 6.0°C during the late October to mid-November surveys in the Illecillewaet and Incomappleaux River systems in 2005 and 2006 (Appendices 4b-c). For the small number of reaches where comparisons could be made, peak summer temperatures recorded on temperature loggers were 1.3 to 2.1°C cooler in 2005 than in 2004 (Table 1). Temperature data were not collected in 2006. In most cases, we were able to conduct surveys when turbidity was low; underwater visibility at the sampling sites was good to excellent, ranging from 4-9 m. Exceptions were sites in the Jordan River in 2004 (1.5 m visibility) and the lowermost site in Caribou Creek in 2004 (2.5 m visibility).

Sites ranged from 20-125 m in length and from 105-3520 m^2 in area (Appendix 1a-c). Stream gradient at individual sites ranged from 0.1% at some sites in the Illecillewaet and Incomappleaux Rivers, to 8% at the uppermost Taite Creek site. Among sites, thalweg depth averaged 0.66 m, and ranged from 0.20 m to 1.9 m. Large woody debris averaged 0.2 m^2/linear m, but was absent from many sites. Maximum LWD density observed was 1.9 m^2/linear m in a run containing a logjam in the Illecillewaet River mainstem.

3.2.1 Mark-recapture estimates of snorkeling efficiency

Mark-recapture data were collected in 2004 and 2005 at a total of 31 sites in five ALR tributaries (Appendices 3a, b). At 23 of these sites bull trout and rainbow trout parr were captured for marking at night by snorkelers, while at the remaining eight sites, daytime electrofishing was used to capture parr. Mark-recapture data from some sites were combined with data from other sites in the same reach because of low numbers of marks deployed, thus, the actual number of site-level mark-recapture estimates of snorkeling efficiency (q) was 17 for bull trout and 14 for rainbow trout (Table 2). Kolmogorov-Smirnov one-sample tests of normality (Zar 1996) for site-level snorkeling efficiency data for both bull trout and rainbow trout in ALR study tributaries (for pooled 2004 and 2005 data) were non-significant for both arcsine square root-transformed data (bull trout: $n = 11$, $P = 0.835$; rainbow trout: $n = 11$, $P = 0.380$) and untransformed data (bull trout: $P = 0.833$; rainbow trout: $P = 0.460$), suggesting that the data could be treated as normally distributed. We pooled data from 2004 and 2005 for all analyses because estimates of mean snorkeling efficiency were not significantly different between years (t-tests; bull trout: $t = 0.94$, $df = 9$, $P = 0.37$; rainbow trout: $t = -1.59$, $df = 9$, $P = 0.15$), but snorkeling efficiency did differ between bull trout and rainbow trout ($t = -3.19$, $df = 20$, $P = 0.005$). We therefore conducted separate analyses for each species.

The assumption of site closure for bull trout and rainbow trout appeared to be met reasonably well at the mark-recapture sites. During snorkeler surveys of mark-recapture sites one night after marking had occurred, relatively high proportions of the number of marks released into the sites were seen (Table 2), and few fish had moved from the marking site to adjacent upstream and downstream sections that were also surveyed (Table 3). Of the total number of marked bull trout that were resighted by snorkelers (87 fish for all sites and methods combined, all were within the original marking site (Table 3). Of the 176 rainbow trout resighted by snorkelers, 12 (7.4%) had moved beyond the original marking site into either the upstream or downstream section (Table 3). The percentage of snorkeling-captured, marked rainbow trout that moved was significantly lower than the percentage of electrofishing-captured, marked fish that moved (6% vs. 23%, respectively; $\chi^2 = 5.12$, $df = 1$, $P = 0.03$; Table 3), but this test had low power owing to small sample sizes.

Intercorrelation among habitat variables measured at mark-recapture sites was high, so several were eliminated prior to conducting logistic regression analyses. Variables eliminated included site length, site area, mean depth, D50, % boulder, cobble, and gravel substrates, % boulder cover, and % deep water cover. Pairwise correlations among the remaining variables in the model (site width, maximum depth, site slope, D90, % fine substrates, overhead cover, turbulence cover, undercut bank cover, large wood debris cover, capture method, and body size) were less than 0.7. An exception to this was site width and site gradient ($r = 0.77$,) but site gradient was retained as a predictor variable because no other variables measured a similar attribute.

In the first logistic regression run body size and stream temperature variables were left out. There was a large amount of missing data in the case of temperature, and size-specific marking occurred in 2005 only. Direct logistic regression analyses for other predictors combined were significant for both bull trout ($P = 0.009$, McFadden's rho-squared = 0.082, $n = 211$) and rainbow trout ($P = 0.050$, McFadden's rho-squared = 0.058, $n = 245$). However, in each case there was only one significant predictor variable, overhead cover in the case of bull trout ($P = 0.037$) and capture method for rainbow trout ($P = 0.008$). When multicolinearity is present, which is evident in our data given the relatively high level of intercorrelation amongst habitat variables, individual predictors may not be significant even when they are highly related to the dependent variable in the population (Tabachnick and Fidell 1996). Therefore, to test the significance of individual predictors we repeated the analysis as a stepwise logistic regression employing a backward elimination procedure recommended for the situation of multicolinearity (Zar 1996). For both species, capture method (snorkeling versus electrofishing) was the best predictor of the likelihood of individual marked fish being seen (bull trout: $P = 0.001$; rainbow trout: $P = 0.002$). No other predictors were significant in the stepwise regression for rainbow trout, although overhead cover remained a significant predictor of the likelihood of whether a marked bull trout was seen ($P = 0.029$).

Estimated mean snorkeling efficiencies for bull trout and rainbow trout were twice as high at sites where snorkelers captured fish for marking (bull trout: 48%; rainbow trout: 72%; Table 2) compared to that at sites where electrofishing crews captured fish for marking (bull trout: 24%; rainbow trout: 33%; Table 2).

Because of the potential for multicolinearity among habitat variables and marking method category, we re-computed the logistic regression analyses of snorkeling efficiency with habitat variables with the electrofishing-based mark-recapture data removed. Resulting direct logistic regressions were not significant for either rainbow trout (*P* = 0.554, McFadden's rho-squared = 0.030, *n* = 213) or bull trout (*P* = 0.294, McFadden's rho-squared = 0.054, *n* = 144), suggesting that any further significance tests for individual habitat variables would be inappropriate.

To evaluate the statistical significance of stream temperature and body size category variables as predictors of the likelihood of marked fish being seen, we conducted separate logistic regressions for each (snorkeler-marked sites only). Although other habitat variables were included in these analyses as well, we restricted our interpretation to the temperature or body size variable only, as the habitat variables had already been analyzed with the more complete data set. In sites where snorkelers marked fish, stream temperature (*P* = 0.589, *n* = 80) and body size category (*P* = 0.296, *n* = 112) were not significant predictors of bull trout snorkeling efficiency. Temperature was not a significant predictor of rainbow trout snorkeling efficiency (*P* = 0.554, *n* = 166), but rainbow trout less than 100 mm fork length had a significantly lower probability of being seen (stepwise logistic regression *P* = 0.001, *n* = 83) relative to fish of 100-170 mm and fish greater than 170 mm (which were not significantly different from each other).

We also compared snorkeling efficiency among size classes using all data (2004-2005) for snorkeling-marked fish, on the assumption that snorkeler estimates of fish size class in 2004 were accurate (fish were not marked according to size in 2004). Results were similar to those for the 2005 data in that snorkeling efficiency was similar among size categories for bull trout (<110 mm: 48%; >110 mm: 49%) and substantially lower for the smallest size category for rainbow trout (<100 mm: 33%; 100-170 mm: 73%; >170 mm: 64%). For the 2004-2005 dataset, snorkeling efficiency was similar for both size categories of bull trout captured by electrofishing as well, albeit much lower (<110 mm: 22%; >110 mm: 26%). Although data were sparse, 2004-2005 data also suggested that snorkeling efficiency was lower for smaller rainbow trout when electrofishing was used to capture fish for marking (<100 mm: 21%; 100-170 mm: 78%) and, interestingly, more similar to estimates made for the same size categories when snorkeling was the fish capture method (see above).

Because the sample of bull trout captured for marking using the two methods (snorkeling and electrofishing) had similar length-frequency distributions (data pooled for all sites and both years; $\chi^2 = 6.84$, $df = 6$, $P = 0.33$; Figure 2), lower snorkeling efficiency for electrofishing-captured fish cannot be explained by differences in body size of marked fish. During the juvenile surveys, snorkelers noticed that electrofishing-captured fish were frequently partially concealed within the bed material in a manner not typical of unmarked fish. Given this observation, it is more likely that both small and large bull trout modified their behaviour after being electrofished and handled, resulting in lower overall snorkeling efficiency. Our finding of higher snorkeling efficiency for snorkeling-captured versus electrofishing-captured rainbow trout was confounded by fish size: the samples of marked rainbow trout obtained by snorkeling and electrofishing had very different length distributions (chi-square test, $\chi^2 = 28.1$, $df = 3$, $P < 0.001$; Figure 2), with the majority of fish by electrofishing being less than 100 mm. When we considered size classes separately, snorkeling efficiency was similar for snorkeling-captured and electrofishing-captured rainbow trout (<100 mm category: 33% and 21%, respectively; 100-170 mm category: 73% and 75%, respectively; Table 2).

Because we did not find evidence for size-specific snorkeling efficiency for bull trout, and because of our concerns about mark-recapture data from sites where fish were captured by electrofishing, we expanded snorkeler counts of bull trout made during juvenile surveys based on a mean snorkeling efficiency estimate of 48.5%, which was derived from the snorkeling-based mark-recapture data with the two length classes pooled (Table 2). The 95% confidence intervals for q were 22% - 76% for individual sites, and 40% - 57% for the study area as a whole.

For rainbow trout, because of likely size- and gear-related bias as presented above, we used the snorkeling-based mark-recapture data and separate estimates of q for the three size categories (<100 mm: q = 33%; 100-170 mm: q = 73%; >170 mm: q = 64%; Table 2) to expand snorkeler counts at individual sampling sites. We used the pooled data over all size categories to compute a universal 95% confidence interval for the three size classes, with the assumption that error in q for each size class was proportional to the error in q for the pooled size categories (Table 2). The 95% confidence intervals for q were 32% - 98% for individual sites, and 61% - 83% for the study area as a whole.

3.2.2 Length-at-age and age structure

During 2004 and 2005, we collected scale samples from 90 bull trout ranging from 45 mm to 230 mm in length, and from 0+ to 4+ in age (Appendix 2a). We collected 44 samples for rainbow trout ranging from 42 mm to 234 mm in length, and from 0+ to 4+ in age (Appendix 2b). Back calculation of scale ages indicated that a number of the rainbow trout sampled lacked a first-year scale annulus, likely due to late emergence timing and slow first-year growth in these cool streams. Although scale samples were sparse for some tributaries and age classes, estimates of length-at-age for each species derived from scale analysis were comparable (i.e., within 10 mm) to those derived from an independent analysis of frequency histograms of fork length data collected by snorkelers (visual estimates) for the same years. This cross-validation approach suggested that snorkelers could estimate the length of observed fish well enough that fish ages could be reliably determined.

3.2.2.1 Bull trout

For bull trout, in tributaries where sample sizes were adequate to allow for comparisons (n > 5 fish), mean length-at-age was somewhat less in Halfway River and MacDonald Creek (Table 4) compared to Caribou Creek and Greeley Creek. In the glacial Illecillewaet and Incomappleaux systems bull trout had considerably greater length-at-age, but sampling took place more than two months later (late October - mid November versus late August), and extra growth during this period probably explains the difference. Among reaches, mean length ranged from 97-119 mm for age-1+ fish, from 126-169 mm for age-2+ fish, from 175-200 mm for age-3+ fish, and from 254-285 mm for age-4+ and older fish. Mean length-at-age varied little between years in the streams that were sampled in both 2004 and 2005.

In the upstream reaches that contained substantial juvenile bull trout populations (upper Caribou, upper Halfway, upper MacDonald, upper Illecillewaet and tributaries, upper Incomappleaux and tributaries), parr age-structure was similar, with the population consisting almost entirely (90-100%) of age-1+ and age-2+ parr (Figure 3). The proportion of age-2+ bull trout appeared to be higher in Halfway River and MacDonald Creek, where growth rates were lower. In the reaches that were sampled during both years, the proportion of age-1+ declined in 2005 in every reach, particularly in upper MacDonald Creek, where the proportion of age-1+ bull trout in the population declined from 75% to 32%. Low numbers of age-3+ and occasional age-4+ fish were observed in most core bull trout reaches as well (Figure 3, Appendix 6). Age-4+ fish (210-350 mm), presumably adult residents, were limited to the upstream portions of these reaches near the migration barriers. Larger (400-800 mm) adult adfluvial bull trout from the reservoir were also commonly encountered during the late summer juvenile surveys (Appendix 6).

The bull trout population in the upper Tangier River (upstream of the bedrock canyon near the mouth) differed markedly from those inhabiting the other reaches mentioned above, in that age-3+ and older fish represented half the population (Figure 3). This suggests a resident population in contrast to the known adfluvial populations in the other reaches. A series of falls in the canyon near the mouth of the Tangier River likely constitutes a barrier to bull trout migration. In the one site sampled in the 1.2 km long lower reach of the Tangier River below the canyon (site tan1, Appendix 4b), most bull trout observed were age-1+ and age-2+ parr, suggesting an adfluvial population.

3.2.2.2 Rainbow trout

Rainbow trout were somewhat smaller for a given age than bull trout, especially in the case of age-0+ and age-1+ fish (Table 4). Similar to the trend for bull trout, rainbow trout were largest for their age in the Illecillewaet and Incomappleaux Rivers and Caribou Creek, and smallest in Halfway River, with other tributaries having intermediate length-at-age values. Among reaches, mean length ranged from 87-114 mm for age-1+ fish, from 115-158 mm for age-2+ fish, from 148-195 mm for age-3+ fish, and from 185-247 mm for age-4+ and older fish. For rainbow trout mean length-at-age varied little between years in the streams that were sampled in both 2004 and 2005.

Juvenile rainbow trout age-structure varied considerably among reaches, exhibiting a strong pattern of predominately younger fish in non-glacial reaches immediately upstream of the reservoir and not separated from the reservoir by major obstructions (Burton, Taite, lower Kuskanax, lower MacDonald; lower Halfway, lower Caribou), and a greater diversity of ages (age-1+ to age-4+ or older) in glacial reaches (Illecillewaet and lower Incomappleaux rivers and their tributaries) and those separated from the reservoir by major obstructions (upper Halfway, upper Caribou, Greeley, Illecillewaet) or lengthy sections of high gradient (upper MacDonald). In reaches near the reservoir, 86%-100% of fish (excluding age-0+ fry) were age-1+ or age-2+ (Figure 3), whereas in upper reaches, only 45%-70% of fish were age-1+ or age-2+. Upper Caribou and upper MacDonald Creeks were exceptions to this in 2004 (87% and 93% of fish were age-1+ or age-2+, respectively). Similar to bull trout, in the reaches that were sampled during both years, the proportion of age-1+ rainbow trout declined in 2005 (Halfway River was the one exception; Figure 3), with MacDonald Creek again exhibiting the largest decline (86% in 2004 versus 5% in 2005). Rainbow trout larger than 230 mm in length were relatively scarce in the study tributaries, and were observed mainly in the Illecillewaet and Halfway Rivers, and upper MacDonald and upper Caribou Creeks.

3.2.3 Distribution and abundance

<u>3.2.3.1 Bull trout</u>

Estimates of both linear and areal bull trout density by age-class are summarized for individual sites in Appendix 4. Bull trout parr (age-1+ to age-3+ combined) were observed at very low density (<10 fish/100 m or 1 fish/100 m^2) in 11 of 24 reaches sampled (Figure 4a, Table 5). The highest parr densities (12.6-101 parr/100 m or 1.1-10.2 parr/100 m^2) always occurred in reaches separated from the reservoir either by major obstructions or lengthy sections of high gradient channel (upper Caribou, upper

MacDonald, upper Halfway, upper Illecillewaet and upper Incomappleaux and their accessible tributaries). These were also the coldest reaches sampled, as indicated by spot temperatures recorded during sampling (Appendix 4), by thermograph records for selected streams (Table 1), or by obvious glacial influence.

For each of the four core bull trout rearing reaches sampled in both 2004 and 2005 (upper Caribou, upper MacDonald, upper Halfway, Greeley), parr density varied little between years (Figure 4a). In the lower reaches of the Halfway River and Caribou and MacDonald Creeks, however, parr density was noticeably greater in 2005. In each case this was the result of higher counts of fish in the uppermost site(s) in the reach, nearest the downstream boundary of the upper reach (Appendix 4), suggesting that the distribution of juvenile bull trout in these streams had expanded downstream from 2004 to 2005. This apparent expansion in distribution was associated with cooler temperatures for that year, as indicated by 2004 and 2005 thermograph records from Burton, Caribou, and MacDonald Creeks (Table 1).

The order of importance among ALR tributary rearing areas for bull trout, as suggested by approximated parr (age-1+ to age-3+ combined) standing stocks, was upper Illecillewaet River (23,276 parr in mainstem and tributaries combined; 48.3% of the overall standing stock in sampled reaches; Table 5), followed by Incomappleaux River (12,524 parr; 26.0% standing stock), Halfway River (6,376 parr; 13.2% standing stock), MacDonald Creek (3,256 parr; 6.8% standing stock), and Caribou Creek (2,089 parr; 4.3% standing stock).

3.2.3.2 Rainbow trout

Estimates of both linear and areal rainbow trout density by age-class are summarized for individual sites in Appendix 5. In contrast to bull trout, rainbow trout were absent or in very low abundance in glacially influenced ALR tributary reaches: no rainbow trout were observed in the Jordan River, Albert Creek or Tangier River, or the entire upper Incomappleaux system including tributaries (Table 6; Figure 4b). In the Illecillewaet River and its tributary Greeley Creek, rainbow trout were present at very low densities

(Table 6: <10 fish/100 m or 1 fish/100 m^2). In glacial tributaries (Illecillewaet, Incomappleaux, and Jordan) rainbow trout were scarce even at sites near the reservoir and below any major obstruction (Table 6; Figure 4b; Appendix 5).

In non-glacial reaches separated from the reservoir by major obstructions or lengthy sections of high gradient channel (upper MacDonald, upper Caribou, upper Halfway), rainbow trout were moderately abundant (20-76 parr/100 m or 2.9-5.6 parr/100m^2; Table 6; Figure 4b), and represented by a diversity of age classes (see section 3.2.2), suggesting stream resident populations. Rainbow trout densities were highest (86-403 parr/100 m or 6.6-21.0 parr/100m^2) in non-glacial reaches near the reservoir and below any major obstruction (lower Caribou, lower Halfway, lower MacDonald, Kuskanax, Taite, Burton). Rearing juvenile rainbow trout in these reaches were mostly younger parr (see section 3.2.2), suggesting adfluvial populations. Overall, in reaches that were sampled in both years, rainbow trout densities decreased in 2005 compared to 2004 (Figure 4b), mainly as a result of lower abundance of age-1+ parr.

3.2.3.3 Bull trout and rainbow trout fry

Although snorkelers counted all bull trout and rainbow trout fry (age-0+) they encountered, as anticipated, fry inhabited very shallow, near-shore areas at night and were difficult for snorkelers to detect. Bull trout fry, in particular, were often observed partially withdrawn into the substrate, suggesting that many fry remained concealed within the substrate at night. Snorkeler counts of fry for these two species should be considered, at best, indicators of presence/absence.

For bull trout, the pattern of fry counts among reaches closely matched the pattern of parr density among reaches: most fry (200 in total for 2004-2006; Appendix 6) were observed in the same reaches (upper Halfway, upper Caribou, upper MacDonald, upper Illecillewaet and tributaries, upper Incomappleaux and tributaries) where parr were relatively abundant. In total, eight bull trout fry were observed by snorkelers in the remaining reaches during 2004-2005. Rainbow trout fry counts also showed a comparable pattern to that of parr abundance, with relatively high fry counts in high density parr reaches near the reservoir (Appendix 6: lower Halfway, lower Caribou, lower MacDonald, Kuskanax; Taite, Burton), and few or no fry observed in reaches where bull trout were relatively abundant.

3.2.3.4 Other species

Rocky mountain whitefish (juveniles and adults) and adult kokanee spawners had a similar distribution within the study in the non-glacial tributaries. Both species were present in all study reaches near the reservoir (Appendix 6), but were absent in non-glacial reaches separated from the reservoir by obstructions or lengthy sections of steep gradient (upper Halfway, upper Caribou, upper Illecillewaet, upper Macdonald). In contrast, mountain whitefish juveniles and adults were relatively common in the upper reaches of the glacial Illecillewaet and Incomappleaux systems, suggesting resident populations. Seven eastern brook trout were observed during the study: six parr (125-155 mm) in the upper reach of Caribou Creek (above the major obstruction), and one adult or subadult (230 mm) at the lowermost site in Jordan River. Longnose dace were observed in very low numbers in several southern tributaries, but only at sites near the reservoir (Burton, Caribou, Kuskanax). Sculpins (not identified to species) were observed in upper Halfway River, Jordan River, Kuskanax Creek and above and below the major obstructions in the Illecillewaet and Incomappleaux rivers. Red-sided shiner (*Richardsonius balteatus*) were observed in the lower reach of Illecillewaet River, and, surprisingly, in the upper Incomappleaux mainstem and Kelly Creek. Kelly Creek is a high gradient, large substrate glacial stream located over 20 km from the reservoir and does not provide any habitat normally associated with this species.

3.3 Adult bull trout survey

3.3.1 Overview of redd surveys

During 2004 and 2005 unusually high precipitation resulted in ALR tributaries experiencing frequent periods of elevated discharge during the normal redd survey period. Despite repeated attempts to conduct surveys in other reaches, in 2004 high discharge and turbidity limited successful redd surveys to Halfway River and Greeley Creek (Table 7; see Decker et al. 2005). Additionally, the survey crew made note that

Burton Creek lacked a suitable access point close enough to the suspected barrier for it to be surveyed efficiently, making it a poor candidate for redd surveys as part of any future monitoring program.

In 2005, a redd survey was completed in MacDonald Creek during Sept 16-17. The early timing of this survey was in response to evidence of early spawning activity during the juvenile survey at the beginning of September. Early spawning timing in MacDonald Creek was confirmed by redd surveys in 2005 and 2006. A large storm event (> 80 mm rainfall in 24 hours) on September 29 increased streamflows several-fold (3.2 to 22.2 m^3/s in Kuskanax Creek, WSC, station 08NE006). Following this storm, continued rainfall resulted in discharge levels in ALR tributaries remaining too high to conduct redd surveys until the end of the survey window (October 15), precluding redd surveys in other tributaries. We did not attempt a redd survey in Illecillewaet River at the time of the juvenile survey in mid November because redds would have been difficult to identify following the storm event in late-September when flow increased from 20 to 145 m^3/s (WSC, station 08ND013). Redd surveys in Incomappleaux River tributaries were pre-empted by a road closure that remained in effect for the entire late fall period.

In 2006 stream flows in ALR tributaries remained relatively low throughout the redd survey period and into November. Redd survey conditions remained ideal, even several weeks after spawning had been completed, owing to the fact that, in the absence of high flows, redds had retained their original shape and were readily identified. In 2006, we were able to complete redd surveys in nearly all reaches were we had previously identified adfluvial bull trout populations, including the large mainstems of the Illecillewaet and Incomappleaux rivers (Table 7). This allowed us to approximate the potential size of the adult population for the reservoir as a whole, and, assuming the distribution of spawning in 2006 was representative of other years, to compare the relative contribution of the various reaches to the overall population.

Redd surveys in non-glacial tributaries in 2006 (Caribou, Kuskanax, MacDonald, Greeley and Halfway) were initiated on September 15 (Table 7). However, spawning was far from complete at that time. The exception was MacDonald Creek, where spawning appeared nearly complete, as was the case in 2005. Redd surveys were completed for the remaining non-glacial tributaries between September 28 and October 2 (Table 7).

Redd surveys in the glacial Incomappleaux and Illecillewaet systems were conducted between October 28 and November 2 (Table 7), after glacial run-off had subsided. Due to the considerable length of the mainstem reaches in these systems, we conducted a helicopter redd survey prior to foot surveys. We then conducted foot surveys only in mainstem sections where redds had been seen from the air (Figures 1b-c, 1f-g) Surveys in tributaries included the entire accessible section below the barrier. The upper reaches of Sable Creek (Incomappleaux tributary) and Woolsey Creek (Illecillewaeit tributary), located above canyons that we were not able to confirm as migration barriers during the reconnaissance flight of 2005, were also surveyed, and the absence of redds there indicted that barriers were likely present downstream.

In the Incomappleaux and Illecillewaet mainstems, helicopter redd counts underestimated redd numbers by 89% and 60%, respectively (Table 7), but appeared to be reliable indicators of the distribution of spawning. No additional redds were observed during juvenile habitat surveys that covered 7.8 km of the Incomappleaux mainstem (in several sections) not included in the foot redd survey. This supported the results of the helicopter survey with respect to the distribution of mainstem spawning.

3.3.2 Distribution of adfluvial bull trout redds within ALR tributaries

As was the case with juvenile bull trout abundances (Table 5), within individual spawning streams the highest abundances of adult bull trout redds were in most cases associated with reaches separated from the reservoir either by major obstructions or lengthy sections of high gradient channel. In MacDonald Creek all spawning activity in 2005 (167 redds; Table 7) took place in the upper reach, while in 2006, 97 of 112 redds were found there. At some point during 2005-2006, two new debris jams formed obstructions in MacDonald Creek, which influenced redd distribution in 2006 considerably. A debris jam located in the upper reach limited the upstream extent of spawning activity relative to 2005 (Figure 1k), and a second jam in the lower reach was associated with several redds located immediately downstream in areas of marginal spawning suitability.

In Halfway River in 2004, 2.4 and 3.2 km sections of the upper reach and a 9.0 km section of the lower reach were not surveyed due to elevated discharge and time constraints. In 2006 the entire system downstream of the headwater fork located 32.5 km from the mouth was surveyed (including 500 m upstream in the north and south forks, but excluding the lowermost 2.4 km where the channel is highly braided and highly unlikely to be used for spawning by bull trout). Numbers of redds for the sections surveyed in both years (sections 1, 2, and 3; Table 7, Figure 1j) were very similar (110 redds in 2004 versus 120 in 2006), and similar proportions of the total redd number were found in the upper reach (2004: 84 of 110 redds; 2006 109 of 141 redds). In both years the average size of redds located in the lower reach below the obstruction at 10 km was unusually small (Table 7), despite the similar visually estimated size for spawners observed in the lower and upper reaches (660 versus 690 mm, respectively). This observation suggests limited spawning substrate, or that some redds were in fact test digs. The clustering of redds in the stream section immediately below the obstruction (Figure 1j) suggests that it constitutes a migration barrier for some fish. Low numbers of resident adult bull trout were observed spawning in the same areas as adfluvial adults. Residents were visually estimated to be of 200 to 250 mm fork length, and the redds they constructed averaged approximately 0.3 m^2.

The upper portion of the Kuskanax Creek reach accessible to adfluvial bull trout is entrenched in a canyon that snorkelers were unable to access during the juvenile survey. Snorkeling sites located in the downstream alluvial portion of the reach indicated an adfluvial rainbow trout population, and the absence of juvenile bull trout (sites ku1 and ku2; Figure 1m; Appendices 4a, 5a). However, a partial redd survey in 2006 in a 1.4 km section immediately below the migration barrier, indicated the presence of an adfluvial bull trout population in this tributary. We observed 37 redds (Table 7, Figure 1m), and presumably missed others in unsurveyed areas downstream.

Caribou Creek, which was surveyed successfully only in 2006, was an exception to the general pattern of spawning activity predominant in upstream reaches. The majority of adult bull trout and redds (31 of 49 redds; Table 7) were observed in the lower reach (Figure 1n). It is possible that, in the low flow conditions of fall 2006, this obstruction interfered with normal bull trout spawning behaviour in Caribou Creek. Redds located in the upper reach were mostly complete and few (11) adults were still present, whereas almost all redds located downstream of the obstruction were still being actively developed and 56 adults were observed. Moreover, redds observed in the lower reach were all clumped within the first few hundred meters immediately downstream of the obstruction.

The majority of bull trout spawning in the Incomappleaux River system took place in mainstem areas (123 of 165 redds; Table 7, Figures 1f-g) of the upper reach, with the lowest redd observed in the mainstem channel being located 1.4 km upstream of the Kelly Creek confluence, approximately 28 km from the reservoir. The low number of redds observed in tributary reaches is noteworthy. Access to Lexington Creek (0 redds) appeared to be cut off at the mouth, and an unstable debris jam that re-routed Sable Creek (1 redd) across a logging road and into a drainage ditch in 2005-2006 may have diverted fish away from this otherwise suitable-looking stream. Juvenile bull trout densities in Sable were high relative to other Incomappleaux tributaries (Appendix 4c), suggesting greater spawner numbers in years prior to the formation of the debris jam. Menhinick, Boyd and Kelly creeks all exhibited evidence of substantial channel instability, to which previous stream-side forest harvesting appeared to have been a contributing factor.

Similar to the Incomappleaux system, the majority of spawning activity in the Illecillewaet River system took place in the upper mainstem reach (296 of 449 redds; Table 7, Figure 1b-c), and no redds were observed in the lower reach. In contrast to Incomappleaux River, spawning activity in Illecillewaet River was distributed along the entire length of the upper mainstem reach, but spawning did become more concentrated towards the top of the reach, particularly in the vicinity of the Albert Creek confluence. The helicopter survey indicated that redds were absent (or nearly so) in two highly confined mainstem sections with reduced channel widths and larger substrates: an 8.4 km section located with West Twin Creek roughly in its center (Figures 1b-c), and a 4.5 km section immediately upstream of the obstruction near the reservoir.

Albert Creek was the most important spawning tributary for bull trout in Illecillewaet River (87 redds; Table 7), followed by Greeley (48 redds) and Woolsey (18 redds) creeks. The lower Tangier River and West Twin Creek had short sections accessible to bull trout spawners but were not surveyed, so the total tributary contribution for the system will have been underestimated slightly, although West Twin Creek appeared to contain little spawning habitat. The 2006 redd count in Greeley Creek exceeded that in 2004 (48 redds versus 14). This may be partly due to the break-up of a debris jam that was located 400-500 m below the barrier in 2004, above which redds were abundant in 2006, but may also have been partly attributable to a high degree of channel complexity in the lower end of Greeley Creek, and the possibility that a channel that contained a number of redds near the Illecillewaet confluence in 2006 was missed by the 2004 survey crew.

3.3.3 Relative abundance of adfluvial bull trout among ALR tributaries

It is likely that the redd surveys conducted in ALR tributaries in 2006 included the large majority of spawning areas utilized by ALR bull trout. We would argue that the results of these surveys provide a useful index of the relative importance of each major spawning tributary. Other tributaries to the reservoir that are known to support adfluvial bull trout (e.g., Hill Creek) or that may support bull trout (e.g., Burton Creek) are likely of lesser importance. In 2006 we observed a total of 953 redds in the six drainages surveyed. The Illecillewaet River system is clearly the most important bull trout spawning and rearing tributary in the ALR system, with nearly half (449 redds or 47% of the total; Table 8) of the total number of redd observations. When the redd total for Illecillewaet River is combined with that from Incomappleaux River, which was the second highest (165 redds or 17% of the total), these two relatively large, glacial systems and their tributaries account for 64% of the total number of redds counted in the ALR in 2006.

In order, the relative contributions of the remaining, non-glacial tributaries in 2006 were 15% (141 redds) for Halfway River, 12% for MacDonald Creek (112 redds), 5% for Caribou Creek (49 redds), and 4% for Kuskanax Creek (37 redds). As mentioned above, spawning activity in Kuskanax Creek was underestimated because it is likely that not all of the spawning area was surveyed. The total contribution of ‘southern’ haplotype stocks, as represented by MacDonald and Caribou Creek redd counts (Latham 2002), is also underestimated in the above proportions. Short sections of Burton and Snow creeks may also be used by adfluvial ALR bull trout (Bray and Mylechreest 1999). Numbers of bull trout juveniles were negligible at juvenile sampling sites in Taite Creek and Jordan River, but the absence of adfluvial populations in these tributaries cannot be ruled out. A small number of bull trout are known to spawn in Hill Creek (Porto and Arndt 2006), and there may still be some adults spawning in McKenzie Creek (McPhail and Murray 1979). Nevertheless, these tributaries have very limited accessible lengths and their contribution to the overall ALR population would be minor. The lack of redd count data from these tributaries would not greatly affect the above proportions.

4.0 DISCUSSION

4.1 Performance of population assessment methods

4.1.1 Reliability of snorkeler counts of juvenile bull trout

Most studies of the effectiveness of snorkeler counts for juvenile salmonids have been conducted in relatively small streams, at sites that could be enclosed with stop nets (e.g., Peterson et al. 2004; Thurow and Schill 1996). We are not aware of other population estimation methodologies that have been developed for use on juvenile bull trout or of any methodologies that have been developed for larger streams or an entire watershed. Our mark-recapture study showed that nighttime snorkeler counts provided a versatile sampling methodology that could be applied to all habitats found in ALR tributaries despite considerable variation in channel width, gradient, substrate size and other characteristics. Overall snorkel counts were reliable predictors of bull trout and rainbow trout abundance in ALR tributaries. For both bull trout and rainbow trout, snorkeling efficiency was consistently high across all sites provided that snorkeling was employed to capture fish for marking rather than electrofishing. Snorkeling efficiency was also unaffected by variation in physical habitat characteristics and stream temperature over the range in these variables represented by the mark-recapture sites. Considering the modest number of mark-recapture sites included in the study (Table 2), error in estimating snorkeling efficiency was quite reasonable at the site level (bull trout: 95% confidence interval for p = 22-76%; rainbow trout: 95% CI = 32%-98%), and particularly for the study area as a whole (bull trout: 95% CI for q = 40-57%; rainbow trout: 95% CI = 61%-83%).

A key assumption of the mark-recapture methodology was that marked and unmarked fish had equal probabilities of being seen by snorkelers the night following marking. In our study, snorkeling efficiency for bull trout[1] was more than double when snorkeling was used to capture fish for marking as opposed to electrofishing (Table 2), suggesting that this assumption was not met for at least one of the two datasets. One possibility is that the snorkeling efficiency for snorkeler-captured bull trout was positively biased because some bull trout in the mark-recapture sites were more active, and as a consequence more likely to be captured by a snorkeler and also to be seen the following night by a snorkeler. Testing for this type of bias was beyond the scope of this study, but results from a similar study of steelhead parr in the Thompson watershed (Hagen et al. 2005; in prep.) suggests that fish marked by snorkelers can provide unbiased estimates of nighttime snorkeling efficiency. In that study two independent calibration methods were used: mark-recapture using snorkelers to capture fish for marking, similar to this study, and three-pass electrofishing removal estimates. After the removal estimates were adjusted for expected negative bias associated with declining capture probabilities (15-

[1] We were unable to make this comparison for rainbow trout because of low samples size for electrofishing and the confounding effect of size-dependent snorkeling efficiency.

25% in ‘good’ estimates of juvenile salmonid abundance, see Bohlin and Cowx 1990), the removal data provided an estimate of snorkeling efficiency (66%-72%) that compared favourably with the mark-recapture estimate of 70% (Hagen et al in prep.).

The more likely explanation for the difference in snorkeling efficiency for snorkeling- and electrofishing-captured bull trout was that bull trout exhibited a cover-seeking response after being captured by electrofishing, resulting in a negatively biased estimate of snorkeling efficiency. Of the few (16) bull trout that were captured by electrofishing, marked and subsequently detected by snorkelers, several were observed partially concealed within the substrate, suggesting that the assumption of no behavioural effects from electrofishing and marking was not reasonable. Consequently, in developing expansion factors for snorkeler counts of bull trout and rainbow trout at the 2004-2006 sampling sites, we used only the snorkeler-capture calibration data.

Thurow et al. (2006) found relatively low snorkeling efficiency for bull trout (mean = 33.2%). It should be noted, however, that the multiple-pass electrofishing estimates used in that study as accurate estimates of abundance had themselves been calibrated by a mark-recapture methodology that relied on electrofishing to capture fish for marking. If bull trout marked in this manner showed an increased tendency to conceal themselves in the substrate, then the subsequent calibration factors based on recaptures of these fish may have been biased low, which would have biased snorkeling efficiency estimates low as well. Mesa and Schreck (1989) found that wild cutthroat trout released after electrofishing showed lethargy and a cover-seeking response and needed at least 24 hours to recover. These authors suggested that their inability to locate a high percentage of marked fish in several stream sections was due to fish seeking uncharacteristically heavy cover, as opposed to movement to other stream sections. Nordwall (1999) found that movement of brown trout out of study sections was up to 14 times greater after electrofishing than in the period preceding it, with upstream movement most prevalent. In our study, lower snorkeling efficiency for electrofishing-captured fish could not be explained by movement of fish after electrofishing. No bull trout initially captured by electrofishing or snorkeling were found outside the marking section.

Although the proportion of electrofished rainbow trout that moved was higher than the proportion of snorkeler-captured fish (Table 3), the size of the sample was too low to make much inference.

In our study, the sampling protocol for snorkeling-captured fish was designed to minimize the behavioural effects on fish from handling and marking. Fish were captured in a relatively low impact manner (hand nets), were not anaesthetized prior to marking, were released into the same location that they had been captured from (or first seen in), and were allowed a 24-hour recovery period prior to the snorkeler count. Snorkelers noted that, after 24 hours, marked fish occupied comparable locations to unmarked ones and behaved in a similar way.

A second assumption of our mark-recapture methodology was that the populations were closed between marking and resighting events. While our sites were not enclosed, we treated the fish populations within as being closed over the 24-hour period between marking and the snorkeler survey. The incidence of parr emigrating from the mark-recapture sites appeared to be relatively low, consistent with our assumption of site closure. Of the 71 bull trout marked by snorkelers that were re-sighted, 100% had remained within the site, while only nine of 163 (5.5%) rainbow trout marked by snorkelers had moved to an adjacent upstream or downstream stream section. Moreover, snorkelers noted that the majority of the marked rainbow trout that had moved from the original marking site had remained within a short distance of the original site boundaries. We could not detect the number of fish that had moved beyond the adjacent sections, but given that mark-recovery rates within the original sites were relatively high, and that we expanded the sites during the recapture survey to account for small-scale movement, a significant violation of the assumption of population closure was unlikely.

In comparison to our snorkeling efficiency results, Mullner and Hubert (1998) found a relatively more precise relationship between daytime snorkeler counts and electrofishing removal estimates of rainbow trout juveniles ($R^2 = 0.95$) than we did for nighttime snorkeler counts versus Peterson mark-recapture population estimates for bull trout or rainbow trout (R^2 = 0.67 and 0.70, respectively). Estimated snorkeling efficiency for rainbow trout in their study (65%) was comparable to our mark-recapture estimate of snorkeling efficiency for the same species in ALR tributaries (72% for pooled size categories). The nighttime snorkeling efficiency recorded by Hagen et al. (in prep.) for Thompson River system tributaries (70%) is also comparable to the ALR mean for rainbow trout. Snorkeling efficiency in other studies has been more variable. Daytime snorkeling efficiencies relative to electrofishing of 103% and 102% have been recorded for steelhead (Hankin and Reeves 1988), and cutthroat trout (Griffith 1981), respectively, and nighttime snorkeling efficiencies of 75% and 33% relative to removal electrofishing and calibrated removal electrofishing, respectively, have been recorded for bull trout (Thurow and Schill 1996; Thurow et al. 2006 – see above).

We found that snorkeler counts provided a faster and less intrusive alternative to electrofishing for sampling juvenile fish populations in ALR tributaries. One disadvantage, however, of snorkeler counts is that the method does not provide the opportunity to directly measure fish fork length or weight, or to collect scale samples for ageing purposes (we obtained this information by sampling fish captured during the mark-recapture study). Regardless, if the objective is to obtain age-specific estimates of abundance (as was the case in this study), as opposed to accurate size or growth data, our results suggest that visual estimates of fish length made by experienced snorkelers are adequate for this purpose. Estimates of length-at-age derived from analysis of histograms of snorkeler-estimated fish lengths were comparable (within 10 mm) to estimates derived from analysis of scale samples collected during the mark-recapture study. Because juvenile bull trout and rainbow trout grow relatively slowly in ALR tributaries, with the result that year classes are relatively well separated with respect to body length, a small degree of error in visual estimation of fish length may not be overly serious for separating age classes. Other researchers have also found that reasonably accurate estimates of fish length can be made when snorkelers are experienced and adjust for underwater magnification either during or after sampling (Griffith 1981; Campbell and Neuner 1985; Thurow and Schill 1996; Mullner and Hubert 1998; Hagen et al. 2005).

4.1.2 Size- and species-related differences in snorkeling efficiency

The mark-recapture study (based on snorkeler-capture of fish for marking) indicated size- and species-related differences in snorkeling efficiency. For age-1+ and older rainbow trout, snorkeling efficiency was highest for larger fish (Table 2). These fish tended to hold further from shore, and were less associated with cover or the stream bottom. Lower snorkeling efficiency for bull trout (q = 42%-50%) was consistent with our observations that bull trout parr of all sizes preferred shallow margin habitat and were more cryptically coloured, making them more difficult to detect than rainbow trout. Unlike rainbow trout, there was no evidence that snorkeling efficiency differed between size classes for bull trout parr. The strong benthic orientation of bull trout during the juvenile life stage has been observed by others, and may be an adaptation to the steep streams utilized by bull trout for rearing or to competition from other salmonids that specialize in feeding from the drift (McPhail and Murray 1979; Nakano et al. 1992; Hagen and Taylor 2001). Snorkeling efficiency was lowest for small (<100 mm) rainbow trout parr (q = 33%-38%). Similar to bull trout, these fish occupied shallow margin habitat at night, although they were sometimes observed in swifter, more turbulent water that was difficult to survey from above or under the water surface.

The size distribution of rainbow trout in the snorkeler-captured marked population was also affected by lower snorkeling efficiency for smaller parr. Since snorkelers were less likely to see small rainbow trout, they were also less likely to capture them, as evidenced by the truncated mode for age-1+ parr in the length frequency plot for snorkeler-captured rainbow trout (Figure 2). The opposite situation occurred when electrofishing was used to capture fish: the marked population was comprised mostly of parr less than 110 mm in length, with almost no fish greater than 150 mm captured. If we had used either marked population to calibrate snorkeler counts for rainbow trout without marking fish according to size class, snorkeling efficiency estimates would have been biased because capture efficiency differed significantly among size classes. For rainbow trout, the large discrepancy in the estimates of snorkeling efficiency derived from the two methods of capturing fish for marking (snorkeling-capture: q = 72%; electrofishing-capture: q = 33%) was likely caused by unrepresentative marked populations. In contrast to rainbow trout, the marked populations of bull trout collected by snorkelers and electrofishing crews had nearly identical size distributions, suggesting that both methods were suitable for collecting representative size data for this species, although the probability of re-sighting fish captured by electrofishing appeared to be much lower. If additional mark-recapture studies of snorkeling efficiency are to be conducted in ALR tributaries, we recommend that fish, particularly rainbow trout, be marked according to size category.

4.1.3 Logistics of conducting snorkeler counts and electrofishing in ALR tributaries

Despite unusually high precipitation during the 2004 juvenile survey, we found stream flow conditions (discharge, temperature and turbidity) in the non-glacial ALR tributaries in late summer to be highly favourable for conducting snorkeler counts. The juvenile surveys conducted in the glacial Illecillewaet and Incomappleaux systems in November 2005 and October-November 2006 demonstrated that snorkeler counts could be successfully employed in larger, glacial streams as well, once they had cleared. Stream temperatures during these later sampling periods were much colder, ranging from 0.5 to 6.0˚C. We calibrated snorkeling counts from the later sampling periods using mark-recapture data obtained in late summer when stream temperatures ranged from 8 to 13˚C, so we cannot rule out the possibility that juvenile density estimates for the glacial tributaries were biased. Thurow et al. (2006) reported a negative correlation between nighttime snorkeling efficiency and temperature. Conversely, a mark-recapture study of nighttime snorkeling efficiency for steelhead parr indicated no relationship between snorkeling efficiency and temperature (Hagen et al. in prep.).

A major disadvantage of using snorkeler counts as a monitoring tool in ALR tributaries is limited access and difficult sampling conditions. Access points for ALR tributaries are limited; some of the reaches used by bull trout have no good access points at all. Obtaining unbiased estimates of fish density or standing stock requires some form of a random or systematic sampling design, which may not be possible to implement if access is poor. Reaching many of the sampling sites meant travelling considerable distances at night on foot across steep slopes covered in blow-down or logging slash. It should be mentioned, however, that the Halfway, Incomappleaux, and Illecillewaet rivers, which appear to support about 80% of the overall juvenile standing stock, have reasonably good access.

Conducting snorkel counts in smaller, steeper tributaries was quite arduous due to the steepness of the stream channels and the large substrate. In these streams, rather than swimming through the sites, snorkelers frequently had to climb through them, lowering themselves into small pockets of habitat to search for fish. The relatively high snorkeling efficiency estimates reported in this study demonstrate that this type of habitat can be sampled effectively, but only if snorkelers are experienced, physically fit and motivated. With respect to a long-term monitoring program, safety is a concern, even for qualified crews, given the combined risk of night work, fatigue and a hazardous environment.

In glacial streams, cold water and air temperatures in late fall pose additional logistical problems for conducting snorkeler counts. Snorkeling equipment must be suitable for very cold water temperatures (as low as 0.5°C) and in good condition, and caution should be taken to limit site size to prevent snorkelers from becoming dangerously chilled. Weather conditions are also a consideration, and affect the probability of successfully completing the study. Heavy snowfall can bar access to forest service roads (as occurred in 2005), and heavy rain or rain-on-snow events can quickly reduce underwater visibility.

Difficult access and snorkeling conditions and low juvenile bull trout densities both contributed to relatively high costs for completing snorkeler counts in ALR tributaries compared to other streams where we have used this methodology (e.g., Hagen et al. 2005). Low bull trout densities meant that large mark-recapture sites (100 m long) were necessary in order to obtain adequate samples of marked fish. It was necessary to allocate a substantial portion of the resources available for this study to the collection of mark-recapture data for a relatively small number of sites. If long sites are required for the collection of reliable mark-recapture data in the glacial tributaries, it may be difficult to obtain a sufficiently large sample of snorkeling efficiency estimates.

In addition to problems with size-selectivity and fish exhibiting a concealment response following electrofishing (see section 4.1.4), we found that, for the most part, electrofishing was an ineffective method of sampling juvenile fish in ALR tributaries. For one, poor access made transporting electrofishing gear to sites difficult. Secondly, it was not possible to enclose sampling sites bank-to-bank with stop nets even in the smallest study reaches because of the steep, confined nature of ALR tributaries. As a result, during electrofishing it was relatively easy for fish, particularly larger rainbow trout, to escape downstream or to mid channel areas that were too deep and swift to effectively electrofish. Electrofishing capture rates were further reduced by very low conductivity levels (<50 μS/cm) in some of the tributaries. During the mark-recapture study, nighttime snorkelers were able to capture fish at similar, and, in some cases, faster rates than the electrofishing crews. If juvenile surveys are to be conducted in the future, we recommend that electrofishing not be used, particularly not multi-pass removal electrofishing at open or shoreline-enclosed sites for the purpose of estimating abundance.

4.1.4 Adult bull trout survey

Redd counts have been the primary stock assessment tool for adfluvial bull trout, and as such have provided population status information for use in a number of important contexts, ranging from evaluation of population spatial structure and extinction risks (e.g. Rieman and McIntyre 1996), to evaluation of management experiments and system state changes (Chirico and Westover 1998; Bustard 2004), to identification of important natal tributaries and habitat use patterns (Bustard and Schell 2002; Pillipow and Williamson 2004). Despite the widespread application of redd counts, their reliability has only rarely been evaluated quantitatively. Redd counts within watersheds were highly correlated with independent estimates of population size for Dunham et al. (2001) and Al-Chokhachy et al. (2005). However, errors in redd counts must be reasonably low to allow relatively rapid, sensitive detections of changes in the population state (Korman and Higgins 1997; Ham and Pearsons 2000). Errors in redd counts can come from such factors as variability among observers, variation in detection rate among streams, and timing of surveys.

Dunham et al. (2001) found high levels of inter-observer variability in redd counts within particular stream sections that received replicate counts, although Muhlfeld et al. (2006) found that redd detection probability was high and inter-observer variability could be reduced to statistically insignificant levels when all observers were highly experienced. Our study protocols were designed to address this issue. All observers had redd counting experience from other studies, and the entire crew surveyed a reach together at the start of the survey to help standardize their observations. However, budget constraints prevented us from conducting replicate counts, with the exception of stream sections that were also surveyed from the air. Comparison of helicopter and foot surveys showed that the former greatly underestimated redd numbers, but was reliable for determining the general distribution of spawning in mainstem reaches.

In other studies spatial variability in the ratio of redd counts to independent estimates of spawner abundance has been evident. Spawner to redd ratios for the Wigwam River, British Columbia (two years' data; Westover and Conroy 1997; Chirico and Westover 1998), Trestle Creek and East Fork Lightning Creek, Idaho (Dunham et al. 2001), and the Kaslo River, British Columbia (McCubbing and Andrusak 2006) ranged from 1.2 to 2.8 spawners per redd, averaging 2.2 spawners per redd. Our crew conducted the redd counts for the latter study, in which a resistivity counter provided an escapement estimate for comparison with the redd number resulting in an estimate of 2.2 spawners per redd. The deployment of a resistivity counter in each of the core ALR bull trout spawning tributaries, which would take many years unless more than one counter is available, may allow a direct investigation of this source of error in the ALR system.

Redd surveys in the glacial Illecillewaet and Incomappleaux systems likely occurred about four weeks after the completion of spawning. Earlier surveys in these tributaries are not possible due to glacial run-off. The number of missed redds may have been higher in the glacial tributaries relative to the non-glacial ones tributaries that were surveyed at approximately the completion of spawning. In 2006, low flow conditions persisted from the beginning of the spawning period in September to the completion of our surveys in early November. Redds in glacial tributaries remained relatively clean and easy to identify well after the completion of spawning (Kelly Creek in the Incomappleaux system was a notable exception; substantial algae growth was present over some redds, which were identified primarily by their shape).

If redd counts are to be used to monitor the state of the ALR bull trout population, it is important that environmental conditions allow for the completion of surveys in most years. In 2004 and 2005 high flows prevented us from completing surveys in the majority of the study tributaries. However, an analysis of historical streamflow data for Kuskanax Creek (Water Survey of Canada, station 08NE006), a stream which is representative of medium-sized, non-glacial bull trout spawning tributaries in the ALR, suggested that discharge was unusually high in ALR tributaries in fall 2004 and 2005, and that the poor survey conditions we encountered were not representative of most years (Decker et. al 2006). The analysis suggested that high flow conditions could jeopardize the success of redd surveys in 20% of years. Given the inherent difficulty in conducting stream surveys in a high rainfall area, this failure rate is probably acceptable. The frequency of missed surveys in glacial reaches would be higher because redd surveys must be delayed for roughly four weeks following the completion of spawning in early October to allow air temperature to drop and glacial run-off to subside. In the West Kootenays, fall storms and resultant high streamflows are much more common in October than in September (Decker et al. 2006), and this will increase the probability of redds becoming obscured by substrate movement before surveys can occur.

4.2 Patterns of bull trout and rainbow trout abundance

The 2004 to 2006 juvenile population abundance data show a consistent spatial patterns in the distribution and age-class structure of bull trout and rainbow in ALR tributaries that suggests resource partitioning. In non-glacial tributaries bull trout occurred only in very low densities in reaches near the reservoir; these reaches were occupied by high densities of younger (age-1+ and 2+) rainbow trout. Bull trout parr were relatively abundant in only five watersheds, in upper reaches separated from the reservoir by obstructions (upper Halfway, upper Caribou, upper Illecillewaet and tributaries, upper Incomappleaux and tributaries) or sections of high gradient (upper MacDonald). In the glacial Illecillewaet and Incomappleaux systems and their tributaries rainbow trout were nearly absent, likely because they are not well adapted to cold, highly turbid rearing environments (Haas 2001). In other upper reaches, rainbow trout were present at densities that were comparable to those for adfluvial bull trout. The relatively high proportion of older rainbow trout age classes (age-3+ to 4+) in these upper reaches (Figure 3) is consistent with the notion of a resident life history. The domination of lower reaches by high abundances of younger rainbow trout age classes combined with few older individuals suggests adfluvial populations (e.g. Irvine 1978). Lindsay (1977) confirmed the presence of an adfluvial spawning population of rainbow trout in lower Halfway River by operating a partial-span upstream weir near the stream mouth in the spring. Lindsay (1977) captured a total of 205 migrant rainbow trout spawners from the reservoir, ranging in length from 29 to 54 cm (mean = 40 cm). In another study, Lindsay and Seaton (1978) estimated substantial numbers (≈ 1,000-12,000) of rainbow trout fry (0+) emigrating to the reservoir from, among others, Caribou, Burton, Snow and Taite creeks. Our study suggests that habitat segregation is occurring among populations of adfluvial bull trout and adfluvial rainbow trout in the ALR.

In general, the distribution of bull trout redds among the study tributaries followed the distribution of juvenile abundance (Table 8). In both the glacial and non-glacial systems, redd densities were highest in upper reaches that supported relatively high juvenile densities. In reaches below obstructions, where juvenile densities were marginal at best, redds, if present at all, were found only in the immediate vicinity of the obstruction.

Most reviews of bull trout early life history suggest an extended stream residence (1-3 years; e.g. Shepard et al. 1984; Pratt 1992; McPhail and Baxter 1996). In Lake Pend Oreille in Idaho, which is about 200 km southeast of the ALR, Downs et al. (2006) recently found that substantial numbers of age-0 adfluvial bull trout emigrated from a spawning tributary to the lake, but otolith microchemistry suggested that age-0 emigrants did not make a significant contribution to adult returns (in a sample of 47 adults, most of the bull trout entered the lake at age-3 or age-4, and none entered at age-0). The large amounts of energy expended by adfluvial adults to ascend steep ALR tributaries as far as possible to reach upper reaches where water temperatures are cooler and competition with other fish is low is consistent with the notion of an extended stream residence by juveniles. Nevertheless, investigating the possibility of a portion of fry entering the reservoir in their first year of life and ultimately contributing to the adult spawning population may be warranted[1]. A closely related char, the lake trout (*Salvelinus namaycush*), typically spawns on lake shoals; for this species, all juveniles start life in the lake, where they usually seek deeper water within a month of hatching (Scott and Crossman 1973). In the ALR, the deeper waters could provide an abundant food supply in the form of opposum shrimp (*Mysis relicta*). McPhail and Murray (1979) postulated, based on evidence from otoliths of divergent growth rates for fry[2], the possibility that as many as 15% of reservoir-caught adult bull trout may have entered the reservoir before their first winter.

Juvenile abundance data from study reaches where surveys occurred in more than one year suggest that environmental conditions may be an important factor affecting distribution and abundance. Although the pattern of juvenile rainbow and bull trout distribution in the study tributaries was consistent from year to year, a shift in age class structure from 2004 to 2005 was apparent for both species. For both species, in every case except rainbow trout in upper Halfway River, the proportion of age-1+ fish declined in 2005 (Figure 3). This resulted in a consistent trend of lower overall density in 2005 for rainbow trout, but not for bull trout (Figure 4). Coinciding with lower rainbow trout densities in 2005, bull trout parr densities in the rainbow trout-dominated lower reaches of Halfway, Caribou and MacDonald tributaries increased from near zero in 2004 to low values (3-14 parr/100 m) in 2005. Bull trout densities in upper, core reaches, where

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[1] On 19-20 August 2004, reservoir shoreline habitat (690 shoreline m in total) on either side of four tributaries (Halfway, St. Leon, Snow-Burton-Caribou complex, Octopus) was surveyed by snorkeling out to a depth of 3-5 meters, and sampled with baited Gee traps. No bull trout juveniles were sighted or captured, probably as a result of high surface water temperatures (20-22.5 °C) at that time (S. Arndt, CBFWCP file data).

[2] This remains uncertain: an analysis by Burrows (1993) using rainbow trout that had a known time-of-entry into Loon Lake, BC, suggested that using differences in circuli width to distinguish early and late-entry fish in the lake was entirely unreliable.

competition from rainbow trout was presumably less, did not increase in 2005. Apparent higher survival conditions for bull trout in lower reaches in 2005 may have resulted from lower interspecific competition, or a combination of that and lower stream temperatures compared to 2004.

Bull trout distribution in a watershed is often limited such that core rearing habitats represent a small fraction of the total amount of stream habitat available (McPhail and Baxter 1996; Bustard and Schell 2002). This appears to be the case in the ALR tributaries. Temperature and competitive interactions with other species, the outcome of which may be mediated by temperature, have been suggested elsewhere to be important determinants of bull trout distribution and abundance, with most other habitat variables being poorly related. Rainbow trout and bull trout densities were strongly negatively correlated in Goathorn Creek in the Skeena River system (Dave Bustard, Smithers, BC fisheries consultant, data on file), with bull trout dominating in upstream, higher elevation reaches with cooler temperatures. In the nearby Morice River, juvenile bull trout are limited to tributaries that drain high elevation areas associated with glaciers and permanent snowfields, while coho and steelhead dominate low elevation and lake-headed tributaries (Bustard and Schell 2002). Haas (2001) studied bull trout abundance in tributaries of the Columbia River in the Kootenay Region and found that bull trout were not present when summer maximum temperatures were greater than 16°C. Bull trout and rainbow trout parr abundance were also strongly negatively correlated in his study, with bull trout dominating numerically and showing relatively high growth rates and condition factor at sites with summer maximum water temperatures less than 13°C, and rainbow trout dominating at temperatures greater than 14°C. Fraley and Shepard (1989) found few bull trout juveniles in the upper Flathead system, Montana when maximum water temperatures were greater than 15°C, and Saffel and Scarnecchia (1995) found that reaches with high bull trout densities in Idaho streams had summer maximum temperatures of 7.8 to 13.9°C. Dunham et al. (2003) found that at the southern limit of the distribution, temperature was the only biophysical variable that was strongly associated with bull trout presence, and that the probability of occurrence in a reach exceeded 50% when the maximum daily temperature was less than 14-16°C. The notion that water temperature can limit bull trout distribution and abundance is consistent with laboratory trials demonstrating that bull trout have among the lowest upper thermal limits and growth optima of North American salmonids (Selong et al. 2001).

In our study, core bull trout reaches[1] had summer maximum temperatures that were less than the 14-16°C threshold observed in other studies. Lower reaches of ALR tributaries that were dominated by rainbow trout had summer maximum temperatures that exceeded this threshold, ranging from 14.6°C in lower MacDonald Creek to 19.4°C in the lower Halfway River (Table 1).

Our study suggests that the number of core bull trout reaches in non-glacial ALR tributaries is limited. These reaches are highly vulnerable to temperature increases resulting from forest harvesting activities or climate change. The upper reaches of MacDonald and Caribou Creeks, despite their limited accessible length (6.3 km and 1.8 km, respectively), probably contribute a substantial amount of the natural production of the southern ALR stock identified as a genetic unit for conservation by Latham (2002). This underscores the need to conduct additional surveys in reaches not included in this study to determine whether adfluvial bull trout production is occurring in other streams as well, particularly in the southern ALR. Preserving water quality and cool temperatures in core bull trout reaches in non-glacial streams should be considered utmost priorities for conservation and management of this stock. The deployment of thermographs to all identified rearing reaches may be warranted as part of efforts to monitor bull trout persistence. Forest harvesting in drainages containing bull trout should proceed only if increased stream temperatures can be avoided through no-harvest zones in riparian corridors and retention of mature age classes over a substantial portion of the drainage at any one time. The number of stream crossings should also be minimized to reduce the risk of landslides in these steep streams, and to limit access for those seeking to illegally harvest adults as they return to spawn.

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[1] Temperature data were not available for the upper Halfway River, but maximum temperatures were likely < 14°C based on spot temperatures collected during the study. Temperature data were similarly not available for the upper Incomappleaux River, but reaches in this glacial system were likely among the coldest in the study area.

The presence of obstructions in ALR tributaries may also be a factor influencing the relative abundance of bull trout and rainbow trout in these reaches. Migration obstructions occur downstream of most key bull trout rearing reaches identified in this study (upper MacDonald Creek is the exception). The abrupt shift in rainbow trout age-class structure above them suggests that these obstructions constitute barriers to adfluvial rainbow trout. Colder water temperatures and higher flows during the spring probably make it more difficult for rainbow trout to ascend these obstructions than it is for bull trout in late summer. In Gosnell Creek, a tributary to the Morice River, Bustard and Schell (2002) discovered a major migratory bull trout population spawning upstream of a 1.9 m waterfall that was a barrier to the migration of steelhead and coho salmon. Substantial spawning activity immediately downstream of major obstructions in Halfway and Caribou tributaries suggest that they may constitute barriers to some adfluvial bull trout as well. Radio-tagging data suggests that in some cases bull trout are also unable to ascend the obstruction in Incomappleaux River (Bray and Mylechreest 1999).

Although resident rainbow trout are present above obstructions in Halfway and Illecillewaet rivers and Caribou Creek, greater recruitment from highly fecund, adfluvial adults may provide bull trout with the advantage necessary to compete successfully. There is no obvious obstruction limiting the migration of adfluvial rainbow between the upper and lower reaches in MacDonald Creek, but the relatively low density and more diverse age structure of rainbow trout in the upper reach suggests a resident population. If so, limited recruitment for rainbow trout may also be contributing to the success of bull trout in this reach.

We observed the formation or break-up of large debris jams in several ALR tributaries (MacDonald, Greeley, Sable) during the course of the study. In smaller ALR tributaries, which are typically steep and highly confined, debris jams appear to form easily and can become obstructions or even barriers to upstream migration. In Greeley and MacDonald creeks, the break up and formation of debris jams, respectively, resulted in markedly different spawning distributions from one year to the next. Streamflow during spawning migration can also affect whether obstructions are passable. In 2005, a large portion of the adult bull trout spawning population in Caribou Creek expired while trying to ascend a waterfall to gain access to prime spawning and rearing habitat (see section 3.1). The following year the falls were modified to reduce the risk of fish mortality, but late arriving fish were still unable to ascend falls due to low flows. These observations demonstrate the vulnerability of bull trout in ALR tributaries to disturbances to the stream channel, particularly disturbances at points of difficult passage. In November 2006, a major rockslide buried the main logging road into the Incomappleaux River Valley within a canyon gorge that creates a major obstruction to fish passage; the road remains closed at the time of writing. The large volume of material deposited on the road could have a significant effect on the hydraulics of the obstruction if it were cleared from the road directly into the river. The canyon is only 5 km from the reservoir, and virtually all bull trout production occurs upstream of this point. Therefore, utmost care should be taken when repairing the road, including a review of the work plan and on-site monitoring by a professional biologist. Proposals for small run-of-the-river hydroelectric projects in ALR tributaries should also be carefully reviewed by fisheries managers. Sites proposed for such projects are usually located in steep canyons where waterfalls occur. Migration obstructions and barriers for bull trout often occur at these same locations. It is common for a large proportion of a bull trout spawning population to stage for several weeks at the base of an obstruction or barrier. For example, all 59 adults observed in Caribou Creek during the redd survey on September 15 were within 100 m of either the barrier or the major obstruction downstream. The construction of diversion tunnels, head pools and penstocks in canyon sections used as staging areas by adult bull trout may affect their spawning distribution and eventual reproductive success. Latham (2002) points out that deliberate manipulation of barriers to improve passage for adfluvial bull trout may negatively impact isolated resident bull trout populations. Likewise, improving fish passage at existing obstructions may allow adfluvial rainbow trout to access upper reaches that are currently used only by adfluvial bull trout.

With respect to adfluvial rainbow trout populations, little is known regarding spawning locations or the relative importance of individual tributaries in the ALR.

Rainbow trout parr densities in the lower reaches of Halfway River and Caribou Creek, in particular, were exceptionally high, exceeding maximum densities observed during a four-year study of steelhead parr abundance in the lower Thompson River system (Hagen et al. 2005). Earlier work by Lindsay (1977; see section 4.2) indicated that Halfway River was used by relatively small adfluvial rainbow trout from the reservoir. Much larger rainbow trout, likely the progeny of introduced Gerrard rainbow trout from Kootenay Lake are reported in the creel survey, but the spawning location(s) of these fish is unclear. We hypothesize that parr production for adfluvial rainbow trout in the reservoir may be widely distributed among a large number of small to medium size streams, most with limited accessible lengths due to impassable waterfalls, and that these streams may be partitioned among stocks differing in body size, diet, and life history.

The apparent inability of the lower Jordan River to support a substantial rainbow trout population, despite its close proximity to the reservoir, may be the result of glacial influence and resultant cooler water temperatures, or the limited amount of spawning[1] and rearing habitat available within its 3 km accessible length. Adult snorkel counts and juvenile electrofishing surveys conducted in Jordan River during 1980-81 (Mason 1985) clearly indicated an adfluvial bull trout production in this tributary at that time (e.g., 160 adults counted during a snorkel survey, and several hundred fry and parr captured during monthly electrofishing surveys). More recent work documented adult bull trout spawning in the Jordan River and the presence of age-0+ bull trout and rainbow trout in a side-channel near the reservoir (K. Bray, BC Hydro, Revelstoke, pers. comm.). In 2004, we snorkeled 3,100 square metres of habitat in the Jordan River and observed only three bull trout parr. Use of the Jordan River by adfluvial bull trout in the 1980's coincided with nearby dam construction on the Columbia River mainstem that cut off access to upstream spawning tributaries; large numbers of adults were captured at the base of the dam at that time. Colonization of Jordan River appeared initially successful, but the current status of adfluvial bull trout production in Jordan River is uncertain.

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[1] Upstream of reservoir influence, substrate in Jordan River appears to be dominated by large cobbles, boulders and bedrock.

4.3 ALR bull trout production relative to that in other watersheds

Comparing bull trout parr density estimates among published studies is difficult, because of differing sampling methodologies, units for density, and geographic locations. Estimates may be for either the best reaches or whole watersheds, for either resident or adfluvial populations, and may be reported either as reach averages or the range observed among sites. Nonetheless, in general, density data available from long-term abundance monitoring programs suggests that bull trout do not attain high densities even in core parts of their range. The range of density estimates observed in studies we reviewed is summarized in Table 9. Working in tributaries of the Lake Pend d'Oreille drainage of northern Idaho, Saffel and Scarnecchia (1995) reported bull trout densities that ranged from 3.9 to 11.2 parr/100 m^2 in the best rearing tributaries. Sexauer (1984, as cited in McPhail and Baxter 1996) reported mean densities of 0.2 to 1.9 parr/100 m^2 in streams draining the east side of the Cascade Mountains in Washington, and Fraley and Shepard (1989) reported mean densities of 41 to 113 parr/100 linear m in index sites in the Flathead River drainage of Montana.

Studied Canadian populations show a similar range of densities. In Line Creek, a tributary of the Elk River in the southern Rocky Mountains of British Columbia, seven years of average bull trout density estimates ranged from 0.45 to 3.19 parr/100 m^2 (Allan 2001; Table 9), averaging less than 2.0 parr/100 m^2 overall. In rearing tributaries of the Salmo River, a tributary of the Pend d'Oreille River in the Kootenay Region of British Columbia, mean reach densities (3-year averages) ranged from 0.2 to 4.8 parr/100 m^2, with 3-year averages at the best sites ranging from 4.0 to 7.5 parr/100 m^2 (S. Decker, data on file). In Goathorn Creek, a tributary of the Telkwa River in northwest British Columbia's Skeena River system, average char densities (bull trout and Dolly Varden not differentiated) ranged from 3.8 to 4.7 parr/100 m^2 for all sites combined, and from 6.2 to 7.4 parr/100 m^2 for the best upstream reaches (Dave Bustard, Smithers BC fisheries consultant, data on file). In the Thautil River/Gosnell Creek system of the Morice River watershed, also in the Skeena system, mean densities of bull trout parr in core spawning and rearing reaches ranged from about 2 parr/100 m^2 to 8 parr/100 m^2, with densities in the best reaches ranging from about 6 to 8 parr/100 m^2 (Bustard and Schell 2002). Nine-year averages for bull trout in spawning and rearing reaches of the Thutade Lake drainage of northcentral British Columbia's Peace River system ranged from 1.8 to 8.4 parr/100 m^2, with densities in the four best reaches ranging from 7.0 to 8.4/100 m^2 (Bustard 2004). Exceptionally high densities of resident bull trout (juveniles and adults) above a waterfall in East Starr Creek of the Thautil/Gosnell system, Morice watershed (>25 fish/100 m^2, Bustard and Schell 2002) are noteworthy, and suggest that maximum densities observed in adfluvial reaches may not be good predictors of the maximum possible for resident populations.

Average bull trout densities in the upper reaches of MacDonald and Caribou creeks, Halfway River, and mainstem and tributary reaches in the upper Illecillewaet and Incomappleaux systems ranged from 12.6 to 101 parr/100 m (mean = 42.0 ± 10.1 parr/100 m) over the two years of the study, or from 1.1 to 10.2 parr/100 m^2 (overall mean among reaches = 3.9 ± 0.8 parr/100 m^2). Although these density estimates remain uncertain because of the small sample sizes for individual reaches (2-8 sites or composite sites made up of several habitat types), they are within the range reported for 'good' bull trout spawning and rearing reaches studied in other locations in western North America. These reaches should be considered as core rearing habitat for bull trout in the ALR.

4.4 Effects of dam construction on bull trout populations in the ALR

It was previously assumed that several tributary reaches near the reservoir (lower Halfway River, lower Caribou, lower MacDonald, Burton, and Kuskanax creeks) were used primarily by adfluvial bull trout. However, we found that they are used primarily by rainbow trout. As adfluvial rainbow trout are native to the ALR, these lower reaches may never have contributed substantially to bull trout production in the ALR, unless changes to the tributary drainages (e.g., clear-cut logging) have occurred that have resulted in substantial temperature increases in these reaches. The construction of Hugh Keenleyside Dam may therefore not have had a large impact on juvenile bull trout production within the immediate ALR catchment, whereas the opposite may be true for rainbow trout. Tributaries upstream of Revelstoke Dam, however, have been lost to ALR production (Martin 1976), but not to the Columbia River as a whole. Adult bull trout were observed and captured in large numbers at the bottom of both the Mica and Revelstoke Dams after they were completed, and tagging of these fish indicated that many were of Arrow Lakes origin (Sebastian et al. 2000). It's likely that adfluvial bull trout populations that formerly spawned and reared in these tributaries, and contributed substantially to recruitment in the Arrow Lakes, have been able to utilize new lacustrine habitat in Mica and Revelstoke Reservoirs. The magnitude of the effect of dam construction on total bull trout abundance and population status in the region will remain uncertain, however, as quantitative stock assessment was not undertaken prior to dam construction.

As compensation for dam construction and loss of tributary habitat in the Columbia River system, migration barriers in the Illecillewaet and Halfway rivers (a dam constructed in 1898 and a natural barrier, respectively) were altered, allowing adfluvial bull trout to access upstream mainstem and tributary habitats, but not other species. In 2006 this habitat accounted for 62% of the total redd count for the study area.

The effects of dam construction along the Columbia River in British Columbia must also be considered in terms of genetic and demographic conservation management. The fragmentation of formerly contiguous bull trout habitats has resulted in the isolation of Columbia River populations. Because re-colonization and the re-introduction of genetic variation from other source areas is no longer possible, loss of genetic variation is a concern if population sizes are reduced to low levels. An important concept in the conservation management of fishes is effective population size (N_e), or the size of a hypothetical stable, randomly-mating population that would have the same rate of gene loss as the real population N. Generally-accepted guidelines suggest that levels of N_e of 50 and 500 are required to minimize the effects of inbreeding depression and maintain adaptive potential, respectively (reviewed in Rieman and Allendorf 2001). Relating N_e to the actual number of breeding adults is not straightforward for bull trout, because of overlapping generations. However, Rieman and Allendorf (2001) estimated N_e for bull trout to be approximately 0.5 to 1.0 times the average number of adults spawning annually, implying that populations for which the evolutionary potential is to be conserved should average 500-1,000 spawners.

The total redd count in 2006 for tributaries supporting the northern ALR genotype (see section 1.0) factored by an empirical average for the number of spawners per redd (2.2; see section 4.1.4) provides a population estimate of 1,742 spawners in 2006, well above conservation minimums suggested by Rieman and Allendorf (2001). A similar estimate for the southern ALR genotype will be biased low because of missed tributaries and reaches that may have contained redds. Expanding the Caribou redd count by a factor of three times is reasonable in order to account for possible redds in Snow and Burton creeks. The 2006 population estimate for the southern genotype would then be 570 spawners. This number would be considered marginal based on Rieman and Allendorf's minimum threshold of 500-1,000 spawners. Because southern, non-glacial tributaries are near the maximum temperature threshold for bull trout as well (see section 4.2), special management of these watersheds is warranted.

5.0 CONCLUSIONS AND RECOMMENDATIONS

Our results suggested that nighttime snorkeler counts were an efficient way of sampling non-glacial ALR tributaries, and, when calibrated using a mark-recapture methodology, could provide reliable and reasonably precise estimates of abundance for bull trout and rainbow trout parr. When conducted later in the fall after glacial run-off had subsided, snorkeler counts also appeared well suited for sampling fish abundance in considerably larger, glacial tributaries.

We found that redd counts were an effective tool for assessing bull trout spawner abundance in ALR tributaries. Under good conditions, all core spawning reaches in the ALR can readily be surveyed. A near complete count of redds in the study reaches in 2006 provided, for the first time, an estimate of the total size of the spawning population and of the relative contribution of individual tributaries. Flow records for Kuskanax Creek suggest that redd counts could be completed in the non-glacial reaches in most years, despite the poor conditions we encountered in 2004 and 2005. The frequency of missed surveys in glacial reaches would be higher because redd surveys must be delayed for roughly four weeks after the completion of spawning to allow air temperature to drop and glacial run-off to subside.

Inclusion of both juvenile snorkeler counts and adult redd counts as part of a long-term monitoring program would provide valuable stock and recruitment data for bull trout in the ALR. However, if resources were too few to collect both types of data, redd counts should take priority. Access to core bull trout reaches in ALR tributaries is poor, which prevents adequate sampling replication to generate reliable estimates of reach-level juvenile density or standing stock. The advantage of redd counts is that study reaches can be surveyed directly, eliminating the need for sub-sampling and the error associated with spatial variability in abundance. As well, estimates of spawner abundance can be related to the annual creel survey to assess exploitation rates and provide conservation guidelines. Redd counts represent a direct index of the status of the spawner stock in the ALR, and reflect reservoir-specific influences, whereas parr abundance may be governed largely by rearing bottlenecks or environmental conditions specific to the tributaries. We would recommend that annual redd counts be conducted in non-glacial tributaries supporting bull trout populations, and that glacial tributaries should also be surveyed in years when conditions are suitable. Consideration should also be given to conducting redd surveys in tributaries not surveyed in this study that may support bull trout spawning, with priority given to relatively uncommon populations of the southern genotype.

Juvenile abundance data may be most useful for identifying factors that limit the distribution of bull trout rearing within the ALR catchment and in Kootenay region streams in general. In particular, an opportunity exists to strengthen the existing predictive relationship between maximum summer stream temperature and bull trout distribution and abundance, a task that has already been identified as a high priority provincially (Eric Parkinson, MOE Fisheries Research Section, pers. comm.). The difficult part in this process, acquiring reliable juvenile abundance data from a diversity of rearing habitats, has already been completed. We recommend that temperature loggers be deployed in reaches sampled in this study, so that maximum summer temperature data can be acquired and related to mean bull trout and rainbow trout parr densities.

Finally, in weighing the benefits of monitoring bull trout production in ALR tributaries, further attention should be given to the possibility that some ALR bull trout exhibit early life-history strategies other than indicated by our data (i.e, 2-3 years of tributary rearing prior to emigration to the reservoir), as this has implications with respect to whether tributary rearing space represents a bottleneck to adult abundance in the reservoir (McPhail and Murray 1979). As a first step, we recommend that the existing library of adult bull trout otoliths collected from the sport fishery in the reservoir be used to determine whether any of these fish entered the reservoir as age-0 fry. Water samples collected from the reservoir and the major spawning tributaries in 2006 (Appendix 8) indicated sufficient contrast between the tributaries and the reservoir with the respect to the ratios of several elements to calcium (see section 2.5). This provides the opportunity to use otolith microchemistry to estimate age-at-reservoir-entry for adults in the creel sample.

6.0 ACKNOWLEDGMENTS

Steve Arndt of CBFWCP and Karen Bray and James Baxter of BC Hydro were responsible for proposing bull trout abundance monitoring in Arrow Lakes Reservoir tributaries, providing helpful advice and logistical support once the study had begun, and providing a valuable review of the draft. We are indebted to Jeremy Baxter, Jody Schick, Gerry Nellestijn, and Kyle Young for their stalwart efforts in the field. David Bustard's research into bull trout production dynamics in northwestern British Columbia provided background and a valuable comparison with our results. As usual, Dave took a personal interest in our work and gave us access to his research data. Funding for this study was provided by the Columbia Basin Fish and Wildlife Compensation Program, Nelson and BC Hydro, Castlegar. The CBFWCP is a joint initiative of BC Hydro and the British Columbia Ministry of Environment to conserve and enhance fish and wildlife populations affected by BC Hydro dams in the Canadian portion of the Columbia River basin.

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Table 1. Description of biophysical characteristics for 10 tributaries (13 reaches) of the Arrow Lakes Reservoir where juvenile fish sampling was conducted during 2004-2006. Wetted and channel widths were estimated from data collected at juvenile sampling sites.

Stream or reach	Accessible stream length (km)	Mean wetted width (m)	Mean channel width (m)	Ratio of wetted to channel width	Mean annual discharge (m^3/s)	Mean Aug-Sept discharge (m^3/s)	Discharge during juvenile survey (m^3/s) 2004	2005	2006	Peak summer temperature (°C)	Conductivity (µS/cm)
Boyd	1.8	12	35	34%	no gauging stn.	-	no survey	no survey	na	na	na
Burton	6.0 [1]	10.7	19.0	56%	no gauging stn.	-	2.4	no survey	no survey	16.0 (04'), 14.7 (05')	70
Caribou (lower)	4.1	16.2	22.8	71%	no gauging stn.	-	3.8	1.6	no survey	17.5 (04'), 16.4 (05')	100
Caribou (upper)	1.8	14.3	17.3	83%	no gauging stn.	-	na	na	no survey	13.4 (05')	90
Halfway (lower)	9.9	19.0	107.3	18%	no gauging stn.	-	5.0	2.5	no survey	19.4 (04')	40
Halfway (upper)	22.6	17.6	29.8	59%	no gauging stn.	-	no survey	na	no survey	na	30
Illecillewaet (lower)	4.9	26.9	100.7	27%	53.0	64.2	no survey	26.1	14.8	13.6 (05')	150
Illecillewaet (upper)	34.1	21.1	62.4	34%	no gauging stn.	-	-	-	-	na	150
Albert (Illec trib)	2.7	11.5	24	48%	no gauging stn.	-	no survey	no survey	na	na	na
Greely (Illec trib)	1.8	8.7	13.8	63%	no gauging stn.	-	1.1	0.7	no survey	12.6 (97')	70
Tangier (Illec trib)	1.3	24.8	28.0	89%	no gauging stn.	-	na	na	na	na	180
Incomappleaux (lower)	5.0	34.6	54.7	63%	no gauging stn.	-	no survey	no survey	na	na	na
Incomappleaux (upper)	34.6	26.7	110.1	24%	no gauging stn.	-	no survey	no survey	na	na	na
Menhenick (Incom trib)	0.5	7.5	18.0	42%	no gauging stn.	-	no survey	no survey	na	na	na
Pool (Incom trib)	1.2	9.0	13.5	67%	no gauging stn.	-	no survey	no survey	na	na	na
Sable (Incom trib)	1.9	11.4	18.5	61%	no gauging stn.	-	no survey	no survey	na	na	na
Kelly (Incom trib)	1.6	10.2	18.0	56%	no gauging stn.	-	no survey	no survey	na	na	na
Lexington (Incom trib)	0.6	3.6	12.0	30%	no gauging stn.	-	no survey	no survey	na	na	na
Jordan	3.0	29.6	38.2	77%	17.3	15.2	13.5	no survey	no survey	13.7 (00'), 14.4 (01')	na
Kuskanax	8.5	22.2	39.7	56%	14.2	7.6	8.0	no survey	no survey	13.6 (97'), 16.9 (05')	40
Macdonald (lower)	5.7	8.5	13.3	64%	no gauging stn.	-	0.9	0.7	no survey	14.6 (04'), 13.1 (05')	210
Macdonald (upper)	6.3	5.3	10.0	53%	no gauging stn.	-	na	na	no survey	11.7 (05')	170
Taite	2.2	6.3	9.5	67%	no gauging stn.	-	1.1	no survey	no survey	na	40

[1] Barrier was not located during this study, accessible stream length is based on barrier information from previous work (Lindsay and Seaton 1978; Provincial stream inventory data).

Table 2. Summary of mean snorkeling efficiency estimates for bull trout and rainbow trout parr at Arrow Lakes Reservoir tributaries sites in 2004 and 2005. Data are grouped by fish fork length class and by capture method for fish marking. The snorkeler-capture data was further partitioned to reflect different methods of assigning marked fish to a length class during the subsequent snorkeler count (visual estimates by snorkelers in 2004 and 2005 and colour-coded marks in 2005 only). 95% confidence intervals indicate uncertainty in snorkeling efficiency at individual sites. The number of mark-recapture sites is also shown for each capture efficiency dataset.

	Marking method	Method of determining length class of resighted marked fish	Length class (mm)	Fish marked	Marks sighted by snorkelers	Mean snorkeler efficiency	M/R sites	95% CI for snorkeler efficiency
Bull trout	Snorkeling capture	Differential marking by length class (2005 sites only)	<110	48	20	41.7%		
			>110	64	32	50.0%		
			Total	112	52	46.4%	8	
	Snorkeling capture	Snorkler estimate (all sites 2004-05)	<110	58	28	48.3%		
			>110	88	43	48.9%		
			Total	146	71	48.5%	11	21.5% - 76.0%
	Electrofishing capture	Snorkler estimate (all sites 2004-05)	<110	37	8	21.6%		
			>110	31	8	25.8%		
			Total	68	16	23.5%	6	
Rainbow trout	Snorkeling capture	Differential marking by length class (2005 sites only)	<100	8	3	37.5%		
			100-170	44	37	84.1%		
			>170	31	26	83.9%		
			Total	83	66	79.5%	5	
	Snorkeling capture	Snorkler estimate (all sites 2004-05)	<100	63	21	33.3%		
			100-170	131	96	73.3%		
			>170	50	32	64.0%		
			Total	244	149	72.0%	11	31.7% - 98.0%
	Electrofishing capture	Snorkler estimate (all sites 2004-05)	<100	24	5	20.8%		
			100-170	8	6	75.0%		
			>170	1	0	0.0%		
			Total	33	11	33.3%	3	

Table 3. Numbers of marked fish resighted by snorkelers in original marking sites and in adjacent upstream and downstream sections (extended site). Data are organized by species and marking method and are pooled for sites in each category.

	Marking method (bull trout)		Marking method (rainbow trout)	
Resighting location during snorkeler survey	snorkeler capture	electrofishing capture	snorkeler capture	electrofishing capture
Original site	71	16	154	10
Extended site	0	0	9	3
Total	71	16	163	13
Marks resighted beyond original site	0%	0%	6%	23%

Table 4. Summary of estimated mean fork lengths (mm), standard deviations (*SE*) and samples sizes (*n*) for age-0+ to age-4+ bull trout and rainbow trout in Arrow Lakes Reservoir tributaries during 2004-2006. Sample data includes visual estimates made by snorkelers and measurements of fish captured by snorkelers and electrofishing crews.

			0+			**1+**			**2+**			**3+**			**4+ or older**		
	Year	Stream	mean	SE	*n*	mean	*SE*	*n*	mean	*SE*	*n*	mean	*SE*	*n*	mean	*SE*	*n*
Bull trout	2004	Burton				117	-	2	140	-	1						
	2004	Caribou	55	(2.0)	10	106	(1.6)	47	148	(4.0)	5						
	2004	Greely	52	(1.1)	40	109	(1.1)	79	153	(6.5)	6	190	(10.0)	2	210	-	1
	2004	Halfway	49	(3.8)	4	100	(1.6)	43	126	(1.5)	22	170	-	1			
	2004	Jordan				120	(5.8)	3									
	2004	Kuskanax				110	-	2									
	2004	Macdonald	47	(1.0)	23	94	(1.6)	47	136	(3.9)	18				285	(21.8)	5
	2004	Taite	50	-	1	110	-	1									
	2005	Caribou	40	-	2	110	(1.0)	133	150	(1.5)	38	190	-	2			
	2005	Greely	52	(2.7)	10	106	(1.5)	61	157	(2.1)	27	180	-	1	230	-	2
	2005	Halfway				97	(1.2)	36	136	(1.7)	41	179	(3.1)	8	256	(13.4)	12
	2005	Macdonald	49	(1.5)	4	98	(1.7)	40	134	(1.3)	80	175	(1.9)	8	200	-	1
	2005	Illecillewaet	66	(1.0)	13	120	(1.4)	85	161	(2.3)	23	200	(3.2)	5	225	-	2
	2005	Tangier	62	(1.7)	3	135	-	2	169	(4.0)	8	195	-	2	254	(11.5)	7
	2006	Illecillewaet				119	(1.1)	122	155	(1.8)	43	197	(2.6)	10	220	(4.1)	4
	2006	Illec tribs				118	(5.2)	9	169	(4.0)	8	195	-	2	254	(11.5)	7
	2006	Incomappleaux				118	(1.1)	76	153	(2.3)	32	180	(9.5)	6	240	(10.8)	4
	2006	Incom. tribs				116	(1.3)	64	148	(1.5)	23	180	-	2	250	-	2
Rainbow trout	2004	Burton				96	(1.5)	54	130	(2.1)	26				190	-	1
	2004	Caribou	45	(0.8)	38	97	(1.3)	106	146	(2.1)	39	195	-	1	213	(6.1)	6
	2004	Macdonald	41	(1.5)	15	90	(1.5)	54	136	(1.9)	44	177	(3.1)	5	213	(4.4)	6
	2004	Halfway	45	(0.6)	67	92	(0.5)	230	120	(0.6)	158	148	(0.8)	96	185	(2.6)	33
	2004	Greely				98	-	1	115	-	2				220	-	2
	2004	Kuskanax				87	(1.5)	39	126	(1.9)	58	173	(1.6)	8	205	(3.9)	11
	2004	Taite				89	(1.3)	48	130	(1.7)	69	174	(1.5)	14	203	(11.7)	5
	2005	Caribou				104	(1.6)	62	147	(1.6)	89	182	(1.2)	14	213	(2.5)	31
	2005	Macdonald				95	(2.3)	27	140	(1.4)	73	175	(1.3)	16	208	(3.7)	13
	2005	Halfway	65	(1.8)	6	91	(0.7)	171	120	(0.7)	140	148	(0.9)	77	190	(2.1)	63
	2005	Illecillewaet				108	(5.8)	5	158	(3.7)	13	190	-	1	221	(1.4)	7
	2005	Greely				90	(5.8)	3	117	(6.7)	3	170	-	1			
	2006	Illecillewaet				108	(5.8)	5	158	(3.5)	14	183	(3.3)	3	247	(9.4)	15
	2006	Incomappleaux				114	(2.4)	5	157	(6.7)	6	185	-	2	220	(10.0)	3

Table 5. Estimated bull trout parr (age-1+ to 3+) densities in Arrow Lakes Reservoir tributary reaches sampled during 2004-2006. Standing stock estimates are averaged across years for reaches with two years' data. Estimates are based on snorkeler counts adjusted to account for snorkeling efficiency. Too few sites were sampled in individual reaches to allow for meaningful 95% confidence intervals for the estimates.

		fish/100 m			fish/100 m^2			Standing
Tributary	Reach	2004	2005	2006	2004	2005	2006	stock
Burton	Burton	4.5	-	-	0.4	-	-	269
Caribou	lower	1.8	13.9	-	0.1	0.9	-	323
Caribou	upper	101	95.0	-	7.6	8.6	-	1,766
Caribou	**Total**							**2,089**
Halfway	lower	1.0	3.1	-	0.1	0.2	-	203
Halfway	upper	30.8	23.8	-	1.6	2.0	-	6,173
Halfway	**Total**							**6,376**
Illecillwaet	lower mainstem	-	0.0	-	-	0.0	-	0
Illecillwaet	upper mainstem*	-	62.4	*	-	3.1	*	21,282
Illecillwaet	Albert	-	-	13.7	-	-	1.4	370
Illecillwaet	Greely	70.0	80.6	-	8.1	10.2	-	1,318
Illecillwaet	Tangier (lower)	-	23.6	-	-	1.1	-	307
Illecillwaet	Tangier (upper)	-	10.6	-	-	0.4	-	-
Illecillwaet	**Total**							**23,276**
Incomappleaux	lower mainstem	-	-	16.1	-	-	0.3	806
Incomappleaux	upper mainstem	-	-	25.9	-	-	1.6	8,954
Incomappleaux	Boyd	-	-	58.6	-	-	4.9	1,055
Incomappleaux	Kelly	-	-	42.7	-	-	3.7	683
Incomappleaux	Lexington	-	-	3.9	-	-	1.1	23
Incomappleaux	Menhinick	-	-	22.0	-	-	2.9	110
Incomappleaux	Pool	-	-	12.6	-	-	1.5	152
Incomappleaux	Sable	-	-	38.9	-	-	3.4	740
Incomappleaux	**Total**							**12,524**
Jordan	Jordan	7.2	-	-	0.2	-	-	217
Kuskanax	Kuskanax	2.1	-	-	0.1	-	-	183
MacDonald	lower	5.7	9.7	-	0.8	1.4	-	439
MacDonald	upper	43.0	46.5	-	8.2	8.7	-	2,817
MacDonald	**Total**							**3,256**
Taite	Taite	1.5	-	-	0.2	-	-	32

*Data from new sites in 2006 were pooled with 2005 data to estimate mean fish density and standing stock.

Table 6. Estimated rainbow trout (age-1+ to 4+) densities in Arrow Lakes Reservoir tributary reaches sampled during 2004-2006. Standing stock estimates are averaged across years for reaches with two years' data. Estimates are based on snorkeler counts adjusted to account for snorkeling efficiency. Too few sites were sampled in individual reaches to allow for meaningful 95% confidence intervals for the estimates.

		fish/100 m			fish/100 m^2			Standing
Tributary	Reach	2004	2005	2006	2004	2005	2006	stock
Burton	Burton	109.0	-	-	10.1	-	-	3,574
Caribou	lower	212.0	103.3	-	13.1	6.6	-	3,433
Caribou	upper	75.7	51.4	-	5.4	4.7	-	617
Caribou	**Total**							**4,050**
Halfway	lower	403.1	272.8	-	21.0	15.1	-	17,620
Halfway	upper	61.2	41.5	-	3.4	2.9	-	6,164
Halfway	**Total**							**23,784**
Illecillwaet	lower mainstem	-	8.8	-	-	0.3	-	222
Illecillwaet	upper mainstem*	-	8.6	*	-	0.3	*	1,526
Illecillwaet	Albert	-	-	0.0	-	-	0.0	0
Illecillwaet	Greely	2.3	6.2	-	0.3	0.8	-	42
Illecillwaet	Tangier (lower)	-	0.0	-	-	0.0	-	0
Illecillwaet	Tangier (upper)	-	0.0	-	-	0.0	-	0
Illecillwaet	**Total**							**1,790**
Incomappleaux	lower mainstem	-	-	3.5	-	-	0.2	92
Incomappleaux	upper mainstem	-	-	0.0	-	-	0.0	0
Incomappleaux	Boyd	-	-	0.0	-	-	0.0	0
Incomappleaux	Kelly	-	-	0.0	-	-	0.0	0
Incomappleaux	Lexington	-	-	0.0	-	-	0.0	0
Incomappleaux	Menhinick	-	-	0.0	-	-	0.0	0
Incomappleaux	Pool	-	-	0.0	-	-	0.0	0
Incomappleaux	Sable	-	-	0.0	-	-	0.0	0
Incomappleaux	**Total**							**92**
Jordan	Jordan	0.0	-	-	0.0	-	-	0
Kuskanax	Kuskanax	170.0	-	-	8.2	-	-	7,574
MacDonald	lower	88.8	85.5	-	11.5	10.7	-	2,800
MacDonald	upper	29.2	19.8	-	5.6	3.5	-	916
MacDonald	**Total**							**3,716**
Taite	Taite	124.4	-	-	19.5	-	-	1,583

*Data from new sites in 2006 were pooled with 2005 data to estimate mean fish density and standing stock.

Table 7. Summary of redd survey results for Arrow Lakes Reservoir tributaries during 2004-2006. In cases where the entire length of a reach was not surveyed, the actual length surveyed is given in brackets. For 2006, when all reaches were surveyed, the proportion of the total redd count (953) contributed by each tributary or reach is shown.

Tributary or reach	Section	Year	Survey dates	Length (km) [1]	Redds observed	Proportion of 2006 total	Redd density (redds/km)	Mean redd size (m^2)	Live spawners obs.
Halfway (upper)	1	2004	Oct 5-7, 12	12.2	79		6.5	2.4	4
Halfway (upper)	2	2004	Oct-13	4.4	5		1.1	2.4	0
Halfway (lower)	3	2004	Oct 14	2.8	26		9.2	1.2	0
Halfway (lower)	4	2004	Oct 14	1.5	0		0.0	-	0
Halfway (total)	**1-4**	**2004**		**20.9**	**110**		**5.3**	**1.9**	**4**
Greely	all	2004	Oct 2	1.8	14		8.0	1.5	0
MacDonald (upper)	all	2005	Sept 16-17	6.3	167		26.5	1.1	76
MacDonald (lower)	all	2005	Oct 1	5.7	0		0.0	-	0
MacDonald (total)	**all**	**2005**		**12**	**167**		**13.9**		**76**
Halfway (upper)	1	2006	Sept 29-30	12.2	91	0.10	7.4	1.4	20
Halfway (upper)	1.5	2006	Sept 29-30	2.4	12	0.01	5.0	1.4	5
Halfway (upper)	2	2006	Sept 29-30	4.4	2	0.00	0.5	1.3	1
Halfway (upper)	2.5	2006	Sept 29-30	3.2	4	0.00	1.3	1.3	2
Halfway (lower)	3	2006	Sept 29-30	2.8	27	0.03	9.6	0.5	37
Halfway (lower)	3.5	2006	Sept 29-30	3.2	5	0.01	1.6	0.4	2
Halfway (lower)	4	2006	Sept 29-30	5.8 (0)	-	-	-	-	-
Halfway (total)		**2006**		**32.5 (28.2)**	**141**	**0.15**	**4.3**	**1.2**	**67**
Caribou (upper)	all	2006	Sept 15	1.8	0	-	0.0	-	59
Caribou (lower)	all	2006	Sept 15	4.1 (1.1)	0	-	0.0	-	19
Caribou (upper) [1]	all	2006	Sept 28	1.8	18	0.02	10.0	1.1	11
Caribou (lower) [1]	all	2006	Sept 28	4.1 (1.1)	31	0.03	7.6	1.4	56
Caribou (total)	**all**	**2006**		**5.9 (2.9)**	**49**	**0.05**	**8.3**	**1.0**	**67**
Kuskanax [2]	partial	2006	Oct 1	8.5 (1.4)	37	0.04	4.4		36
MacDonald (upper)	all	2006	Sept 16-17	6.3	97	0.10	15.4	1.4	55
MacDonald (lower)	all	2006	Sept 17	5.7 (1.2)	15	0.02	2.6	1.0	20
MacDonald (total)	**all**	**2006**		**12.0 (7.5)**	**112**	**0.12**	**9.3**	**1.0**	**75**
Incomappleaux (upper)	aerial	2006	Oct-29	34.6	13	-		-	0
Incomappleaux (upper)	foot	2006	Oct 28-31	34.6 (13.3)	123	0.13	3.6	-	0
Incomappleaux (lower)	aerial	2006	Oct 28-31	5.0 (0.0)	-	-	-	-	-
Incomappleaux (Kelly)	foot	2006	Oct 28-31	1.6 (1.4)	11	0.01	6.9	-	0
Incomappleaux (Boyd)	foot	2006	Oct 28-31	1.8 (1.3)	18	0.02	10.0	-	0
Incomappleaux (Lexington)	foot	2006	Oct 28-31	0.6 (0.5)	0	0.00	0.0	-	0
Incomappleaux (Sable)	foot	2006	Oct 28-31	1.9	1	0.00	0.5	-	2
Incomappleaux (Menhenick)	foot	2006	Oct 28-31	0.5	0	0.00	0.0	-	0
Incomappleaux (Pool)	foot	2006	Oct 28-31	1.2	12	0.01	10.0	-	0
Incomappleaux (total)	**foot**	**2006**		**47.2**	**165**	**0.17**	**3.5**	**-**	**0**
Illecillewaet (upper)	aerial	2006	Oct-29	34.1 (32.8)	101	-		-	0
Illecillewaet (upper)	foot	2006	Nov 1-2	34.1 (17.0)	296	0.31	8.7	-	0
Illecillewaet (lower)	aerial	2006	Oct-29	4.9 (0)	-	-	-	-	-
Illecillewaet (Albert)	foot	2006	Nov 1-2	2.7	87	0.09	32.2	-	0
Illecillewaet (Woolsey)	foot	2006	Nov 1-2	0.9	18	0.02	21.1	-	0
Illecillewaet (Greeley)	foot	2006	Oct 2	1.8	48	0.05	27.4	-	1
Illecillewaet (total)	**foot**	**2006**		**39.4**	**449**	**0.47**	**11.4**	**-**	**1**

Table 8. Estimated parr standing stocks (averaged for tributaries with more than one years' data) and redd counts (2006 only) for bull trout in study tributaries in Arrow Lakes Reservoir. Also shown are the relative contribution of each tributary to the overall standing stock and redd count. Values for the Illecillewaet and Incomappleaux rivers include numbers from sub-tributaries.

Tributary	Parr standing stock	Proportion of standing stock	Redd count	Proportion of redd count
Illecillewaet	23,276	48.8%	449	47.1%
Incomappleaux	12,524	26.4%	165	17.3%
Halfway	6,376	13.5%	141	14.8%
MacDonald	3,256	6.9%	112	11.8%
Caribou	2,089	4.5%	49	5.1%
Kuskanax	183	na	37	3.9%
Total	47,704	100.1%	953	100.0%

Table 9. Comparison of average bull trout densities in core reaches in Arrow Lakes Reservoir tributaries during 2004-2006 with reach densities reported for bull trout in other streams in western North America.

Drainage	Region	Source	Comment	Range in mean fish density among reaches	
				parr/100m^2	parr/100 m
ALR tributaries	West Kootenay, British Columbia	This study	Core bull trout rearing areas only	1.1 – 10.2	13 – 101
Lake Pend d' Oreille, tributaries,	Northern Idaho	Saffel and Scarnecchia (1995)	Best rearing tributaries	3.9 – 11.2	-
East side Cascades streams	Washington	Sexauer (1984, as cited in McPhail and Baxter 1986)		0.2 – 1.9	-
Flathead River tributaries	Montana	Fraley and Shephard (1989)	Range observed at index sites	-	41 – 113
Line Creek in Elk River drainage	East Kootenay, British Columbia	Allan (2001)		0.45 – 3.19	-
Salmo River, Pend d' Oreille tributary	West Kootenay, British Columbia	S. Decker, data on file	4-8 parr/100m^2 in the best upstream sites	0.2 – 4.8	-
Goathorn Creek, Telkwa drainage	Skeena system, British Columbia	D. Bustard, Smithers, data on file	6-7 parr/100m^2 in the best upstream reaches	6.2 - 7.4	-
Thautil/Gosnell system, Morice watershed	Skeena system, British Columbia	Bustard and Schell (2002)	6-8 parr/100m^2 in the best reaches	2.0 - 8.0	-
East Starr Creek, Morice watershed	Skeena system, British Columbia	Bustard and Schell (2002)	Resident population isolated above waterfall	>25	-
Thutade watershed	Peace River system, Britsh Columbia	Bustard (2004)	7.0-8.4 parr/100m^2 in the best reaches	1.8 - 8.4	-

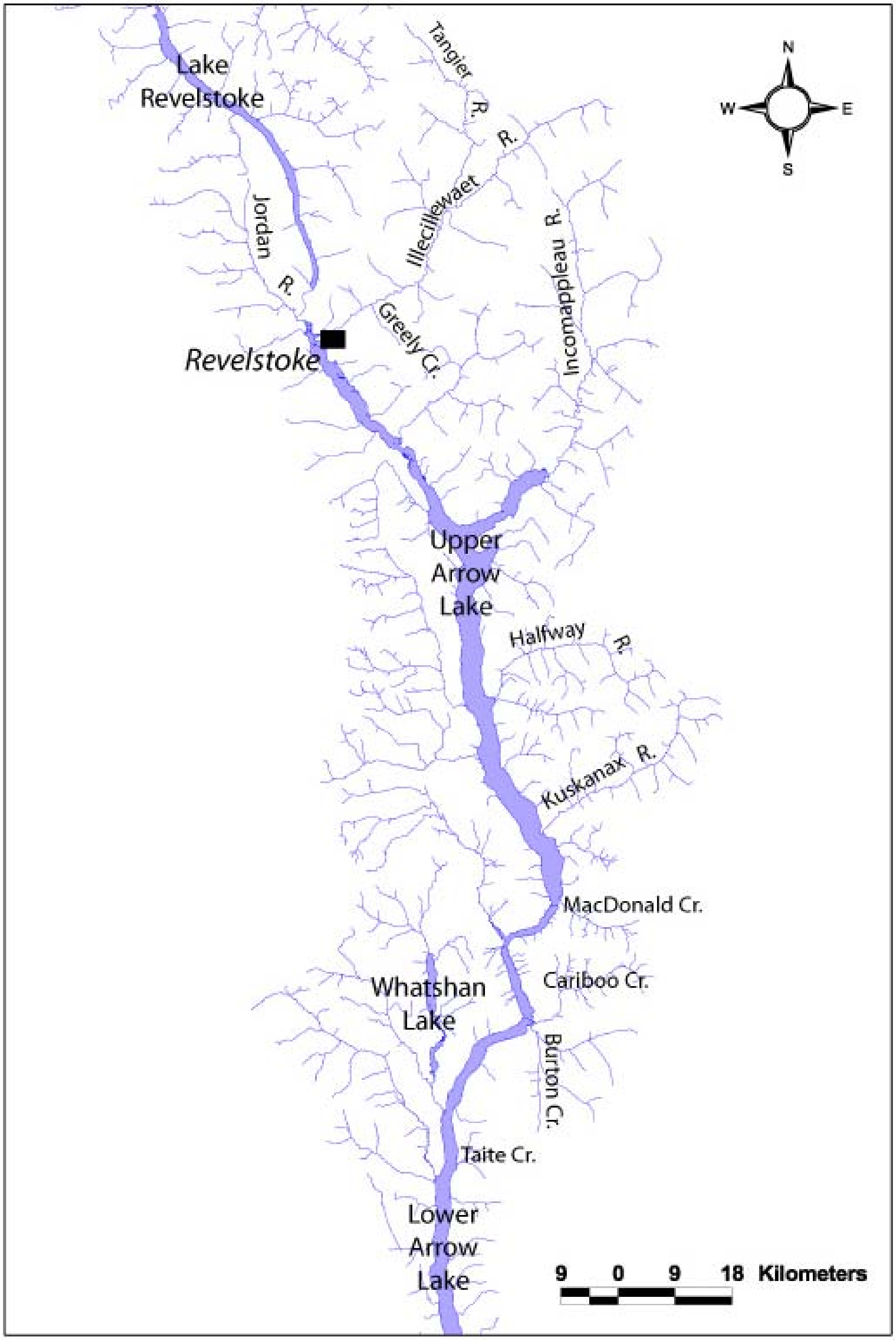


Figure 1a. Overview of the Arrow Lakes Reservoir showing the tributaries included in the study during 2004-2006.

Sensitive Data

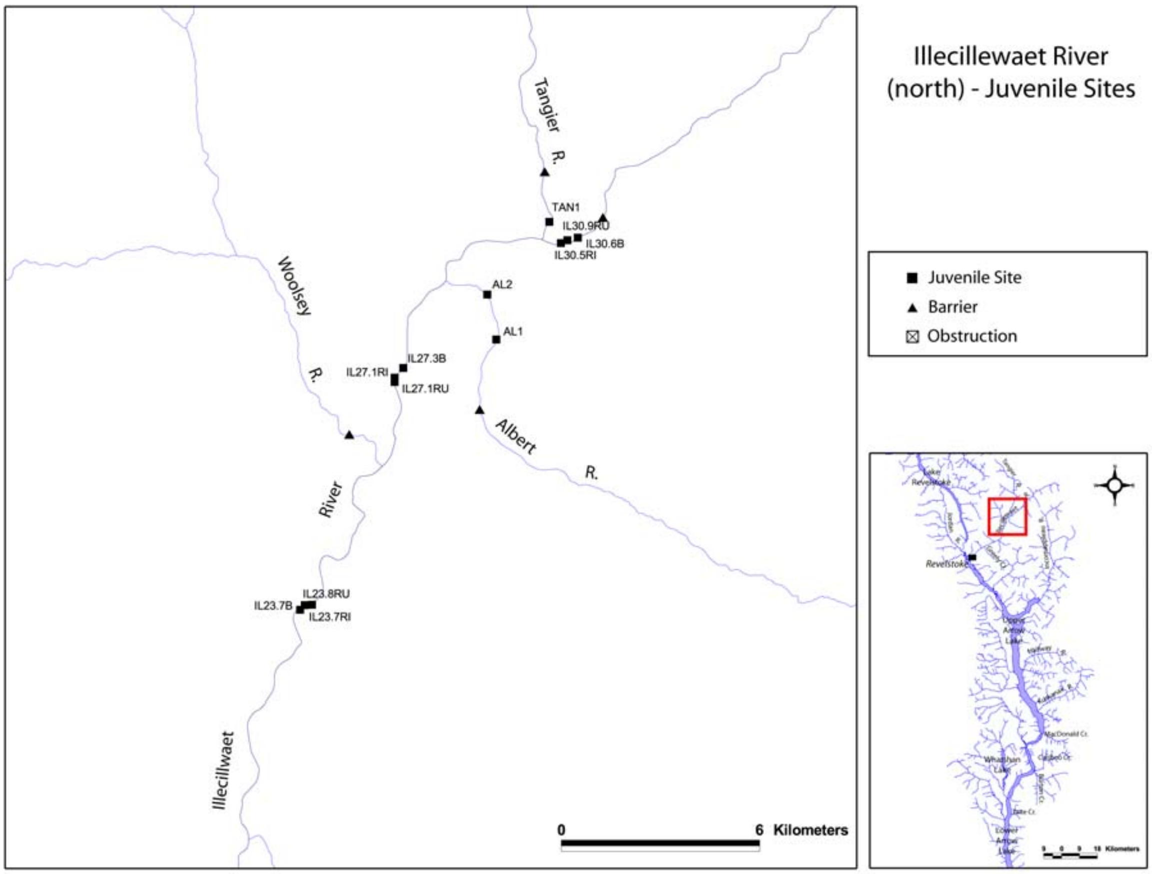


Figure 1d. Upper Illecillewaet River and tributaries with juvenile sampling sites in 2005 and 2006 shown.

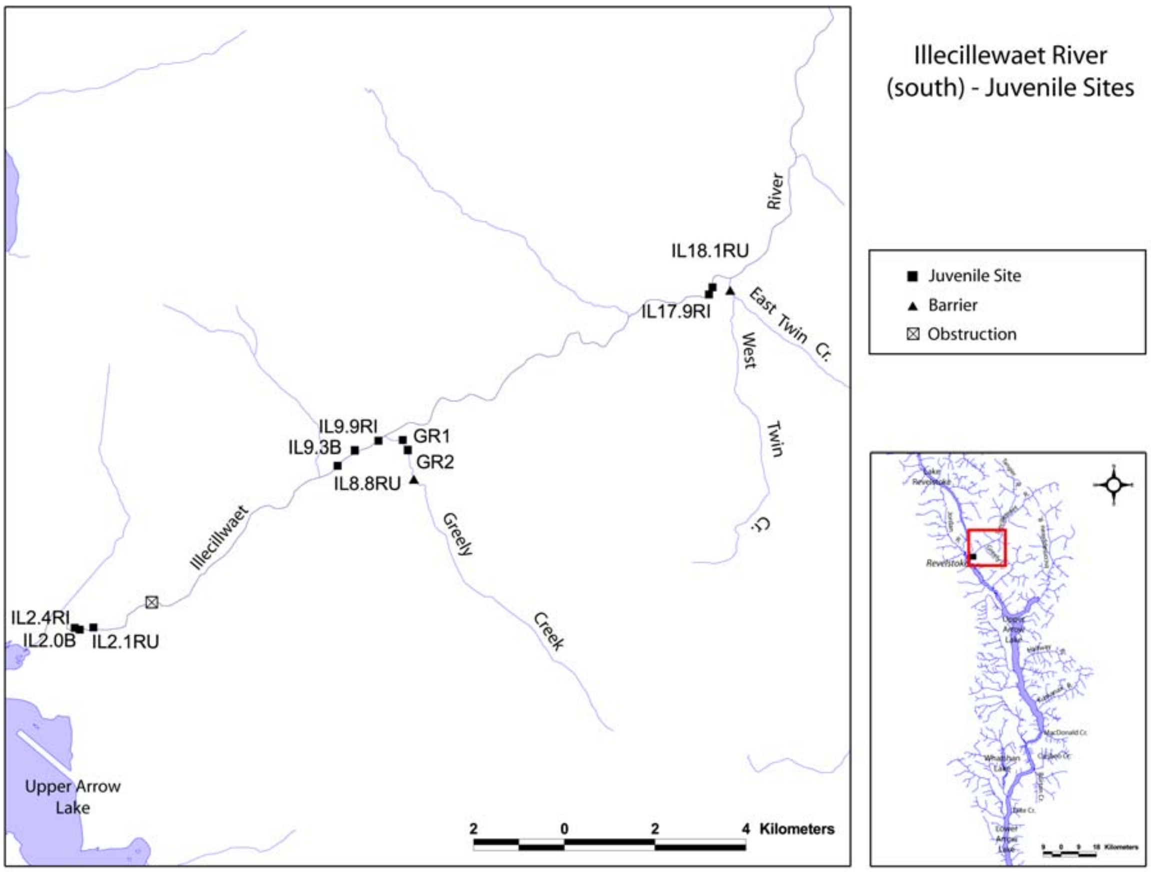


Figure 1e. Lower Illecillewaet River and tributaries with juvenile sampling sites in 2005 and 2006 shown.

Sensitive Data

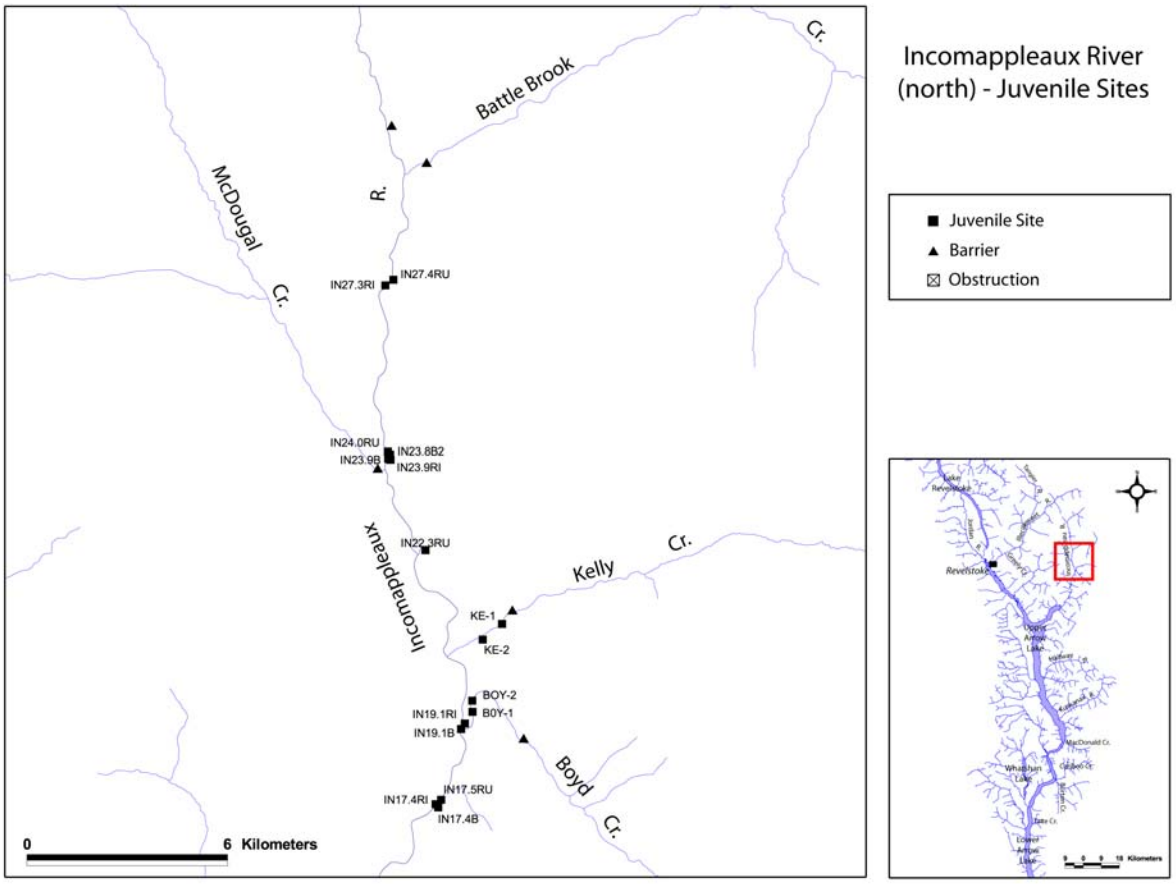


Figure 1h. Upper Incomappleaux River and tributaries with juvenile sampling sites in 2006 shown.

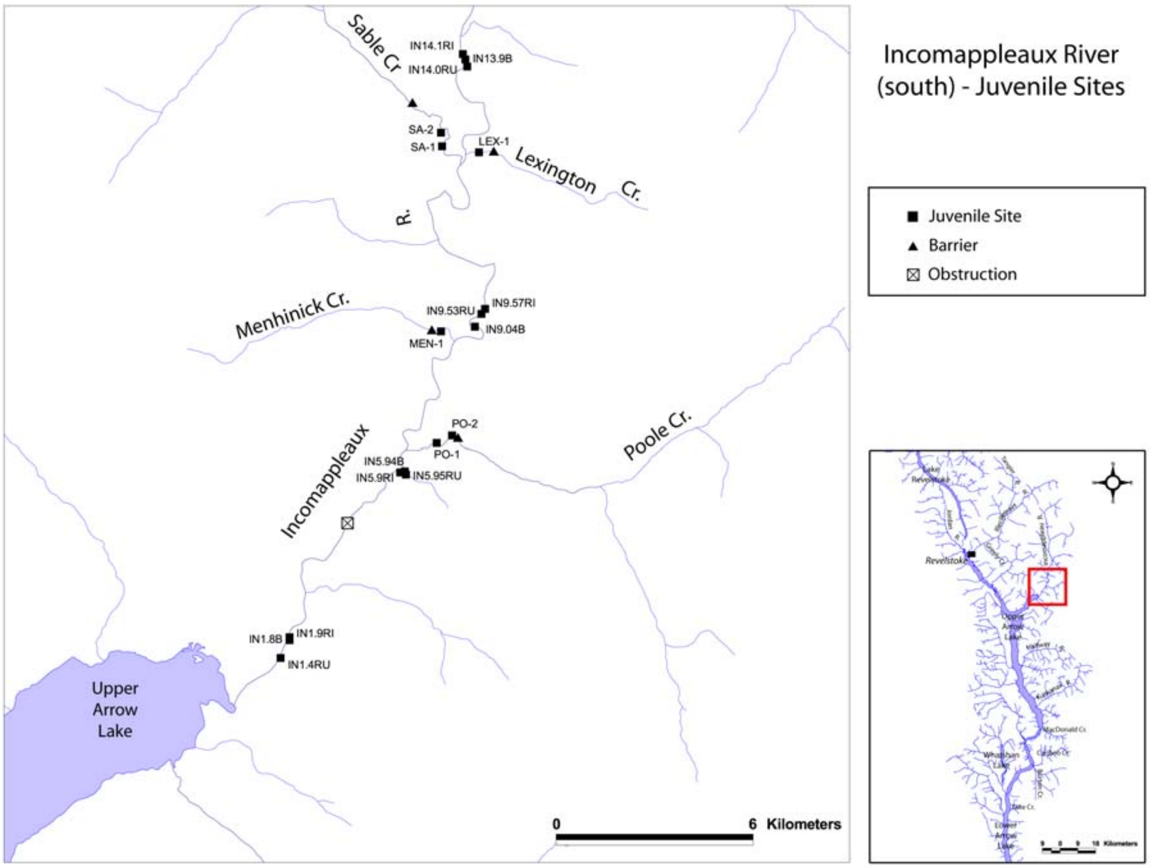


Figure 1i. Lower Incomappleaux River and tributaries with juvenile sampling sites in 2006 shown.

Sensitive Data

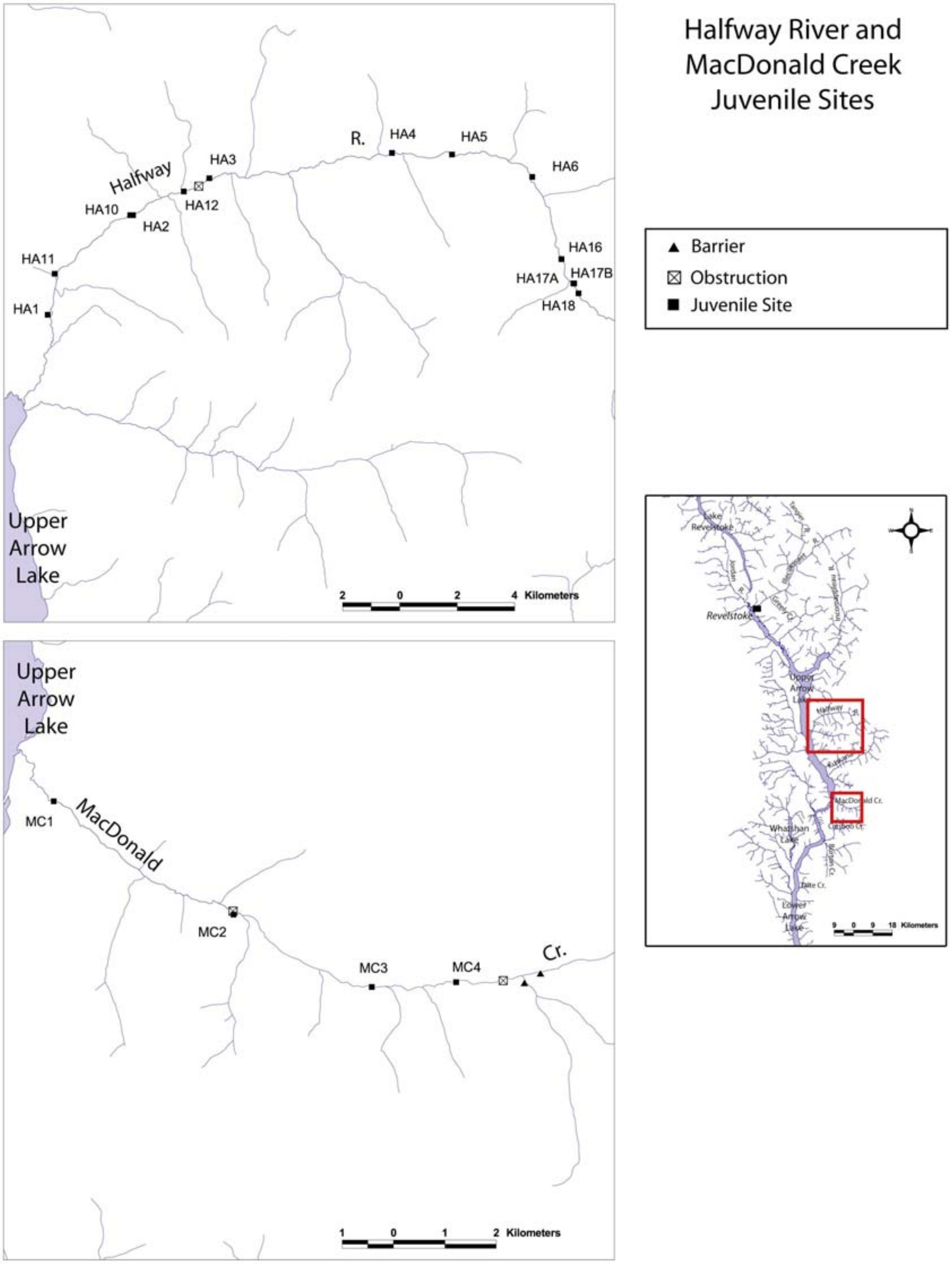


Figure 1l. Halfway River and MacDonald Creek with juvenile sampling locations in 2004 and 2005 shown.

Sensitive Data

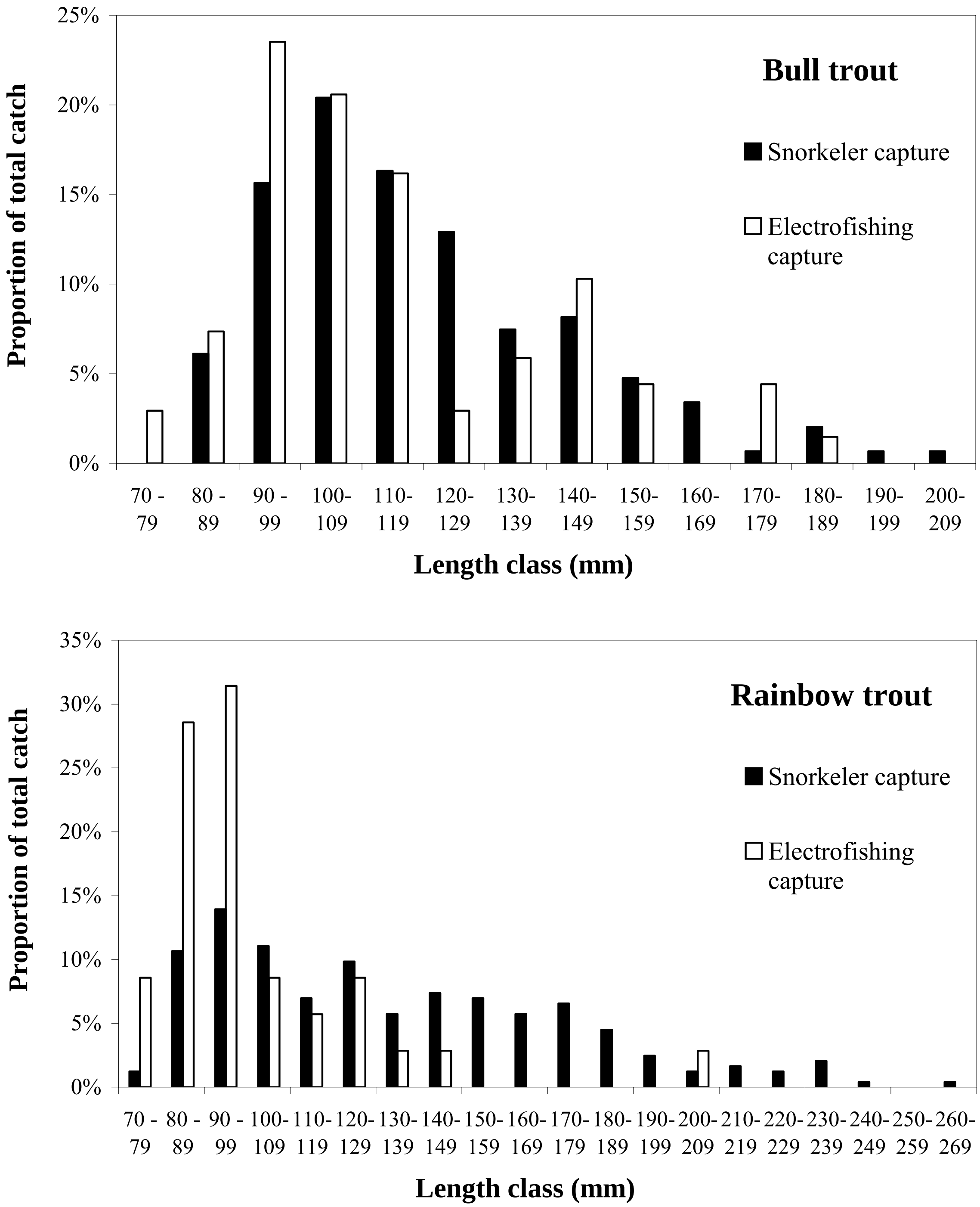
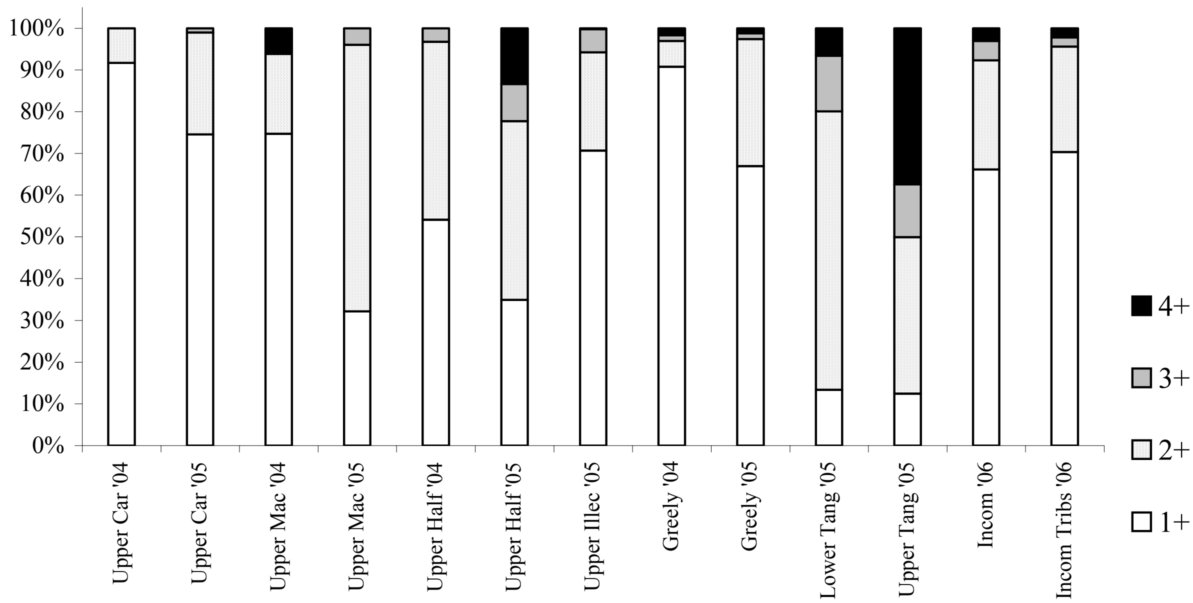


Figure 2. Comparison of fork length distributions (measured lengths) for marked populations of bull trout and rainbow trout collected by snorkelers and electrofishing crews. Data is pooled data for 2004 and 2005.

a) Bull trout



b) Rainbow trout

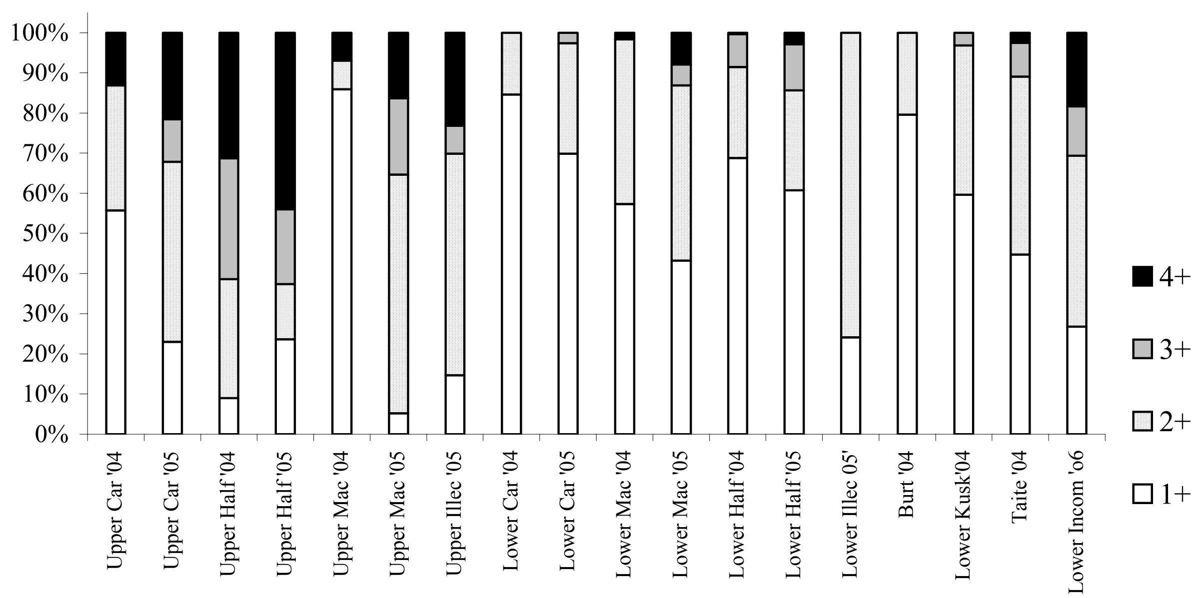
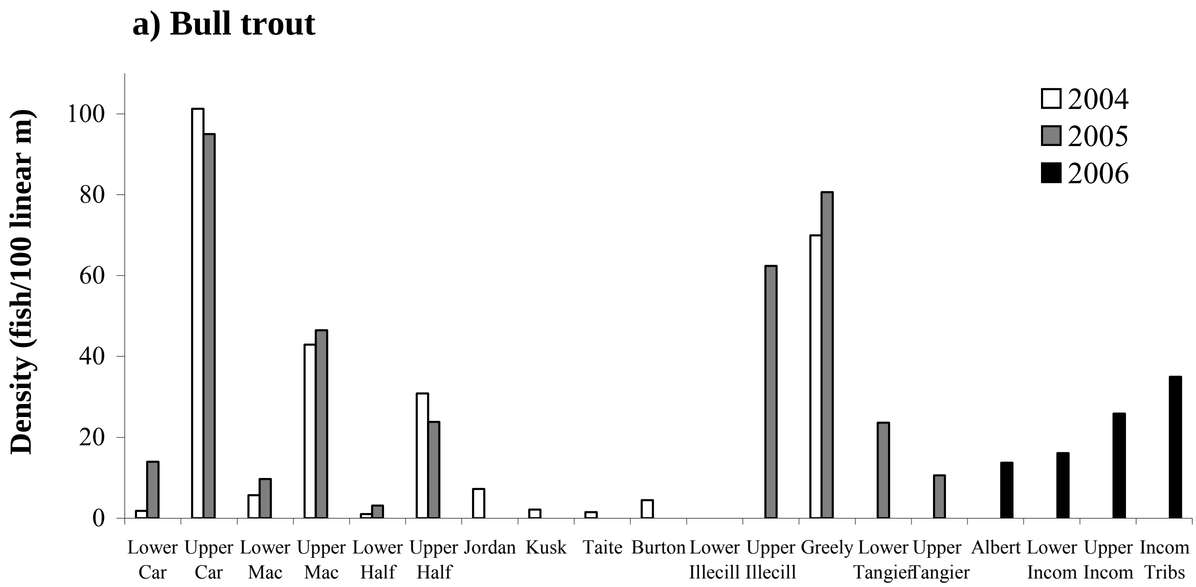


Figure 3. Proportional abundance (% of estimated abundance) for four age classes of bull trout and rainbow trout in Arrow Lakes Reservoir tributary reaches sampled in 2004 and 2005. Age-0+ fish are not included.



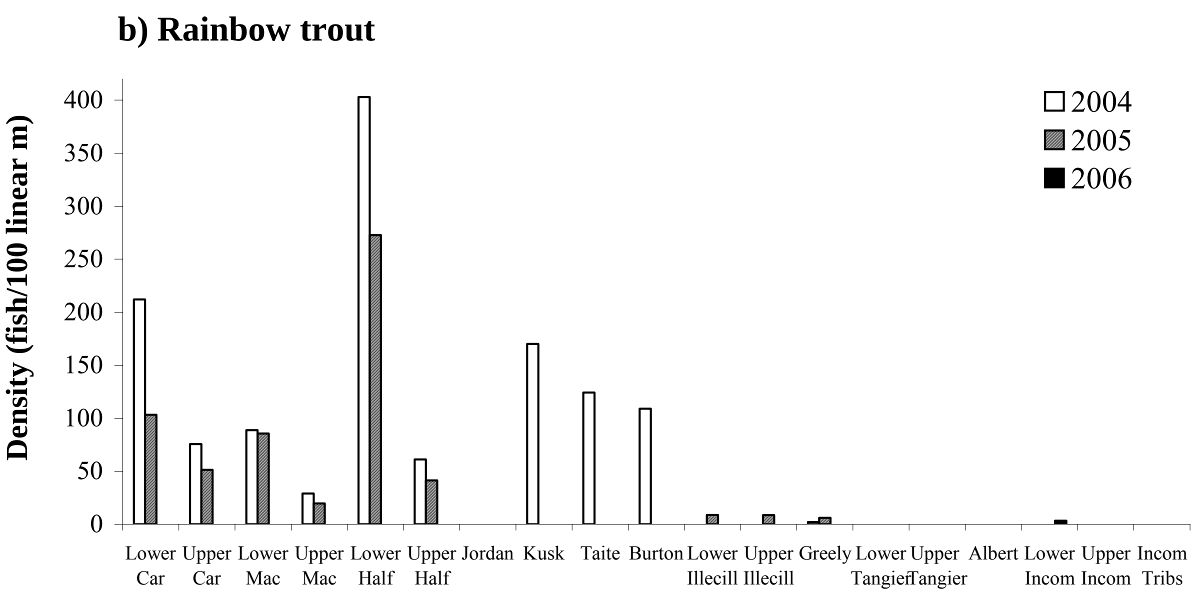


Figure 4. Mean linear densities of bull trout and rainbow trout parr in Arrow Lakes Reservoir tributary reaches during 2004-2006. Note the 4-fold difference in the scale of the vertical axis for the two species. Density estimates are based on snorkeler counts adjusted for snorkeling efficiency. Too few sites were sampled in individual reaches to allow for meaningful 95% confidence intervals for the estimates.

Appendix 1a. Description of habitat characteristics for 29 juvenile sampling sites in 11 tributary reaches of the Arrow Lakes Reservoir in 2004.

Stream	Reach	Site	Year	Habitat type(s)[1]					Substrate compos. (%)				D90 (cm)	D50 (cm)	Fish cover			
				Ri	Ru	P	C	SC	Bd	Cb	Gr	Fi			Turbul e-nce (% site area)	Over-head (% site area)	Under-cut (% bank)	LWD (m^2/ linear m)
Burton		bu1	2004	1	1		1	1	20	45	15	20	55	17	20	5	15	0.37
Burton		bu2	2004	1	1				30	45	10	15	65	18	35	5	10	0.00
Caribou	lower	car1	2004		1		1		35	50	10	5	70	22	30	0	0	0.00
Caribou	lower	car2	2004	1	1				30	50	10	10	50	22	20	0	0	0.05
Caribou	upper	car3	2004	1					40	45	10	15	65	30	20	5	10	0.13
Caribou	upper	car4	2004	2	1	2			40	35	10	15	80	32	20	5	10	0.27
Greely		gr1	2004	4	2		1	2	5	55	30	10	30	12	35	20	10	0.00
Greely		gr2	2004	2	1		2		20	60	15	5	50	20	40	5	10	0.04
Halfway	lower	ha1	2004		2			2	15	75	10	0	40	15	30	0	10	0.00
Halfway	lower	ha11	2004	1				1	30	58	13	0	57	21	20	0	5	0.03
Halfway	lower	ha10	2004	1					0	40	80	10	85	25	40	0	5	0.00
Halfway	lower	ha2	2004	1	1				50	45	5	0	90	25	40	0	0	0.00
Halfway	upper	ha3	2004	1	1				55	40	5	0	60	25	50	0	0	0.00
Halfway	upper	ha4	2004	1	1				25	55	15	5	50	22	30	5	0	0.22
Halfway	upper	ha5	2004	1					25	65	10	0	60	19	40	5	0	0.00
Halfway	upper	ha6	2004	1					30	60	10	0	40	20	40	5	0	0.12
Jordan		jr1	2004		1				5	60	25	10	30	11	10	0	0	0.00
Jordan		jr2	2004	1	1				15	45	25	15	44	14	30	0	0	0.00
Jordan		jr3	2004	1	1			1	40	50	5	5	80	30	40	0	10	0.18
Kuskanax		ku1	2004		1				40	50	10	0	70	25	40	0	0	0.00
Kuskanax		ku2	2004	1					65	30	5	0	90	40	30	0	0	0.00
Kuskanax		ku3	2004	1	1				60	30	5	5	280	38	30	0	0	0.00
Macdld.	lower	mc1	2004	2	2	1	1		5	50	35	10	22	8	20	5	5	0.23
Macdld.	lower	mc2	2004	1	2		2		40	30	15	15	80	25	50	15	40	0.42
Macdld.	upper	mc3	2004	1		1	2		40	40	10	10	55	25	50	35	10	0.18
Macdld.	upper	mc4	2004	2	2		1		10	25	50	15	35	15	30	15	5	0.34
Taite		ta1	2004	1					40	45	10	5	110	23	40	0	0	0.00
Taite		ta2	2004	1					35	50	13	2	70	20	45	5	10	0.00
Taite		ta3	2004	5		5			40	45	13	2	70	23	30	5	0	0.10

[1] Refers to the number of habitat units for each habitat type (riffle, run, pool, cascade, side-channel) occurring in a site.

Appendix 1a (continued)

Stream	Reach	Site	Year	Km from reser-voir	UTM coordinates Zone	Northing	Easting	Site area (m^2)	Site length (m)	Wet-ted width (m)	Chan-nel width (m)	Ratio wetted to chan-nel width	Thal-weg depth (m)	Chan-nel slope (%)	Chan-nel type[1]
Burton		bu1	2004	2.8	11U	436831E	5534084N	396	41	10	17	57%	0.59	2.0%	rp
Burton		bu2	2004	4.1	11U	436781E	5533039N	583	50	12	21	56%	0.64	2.3%	rp
Caribou	lower	car1	2004	0.6	11U	437283E	5536847N	494	38	13	21	62%	0.59	2.4%	rp
Caribou	lower	car2	2004	3.2	11U	439500E	5537086N	1063	55	19	25	78%	0.67	1.0%	rp
Caribou	upper	car3	2004	4.0	11U	440316E	5537602N	720	45	16	20	82%	0.68	2.0%	rp
Caribou	upper	car4	2004	4.6	11U	440820E	5538126N	418	33	13	15	84%	0.83	3.6%	srp
Greeley		gr1	2004	11.0	11U	425457E	5652262N	945	105	9	16	56%	0.39	2.9%	rp
Greeley		gr2	2004	10.9	11U	425533E	5652045N	683	82	8	12	71%	0.40	4.3%	srp
Halfway	lower	ha1	2004	3.3	11U	438075E	5591126N	413	20	21	150	14%	0.83	1.2%	rp
Halfway	lower	ha11	2004	4.9	11U	438498E	5592513N	737	40	19	150	12%	0.61	2.2%	rp
Halfway	lower	ha10	2004	8.0	11U	441371E	5594210N	983	50	20	27	73%	0.77	2.0%	rp
Halfway	lower	ha2	2004	8.4	11U	441470E	5594196N	850	50	17	22	77%	0.82	1.2%	rp
Halfway	upper	ha3	2004	11.3	11U	444271E	5595140N	817	50	16	54	30%	0.79	1.4%	rp
Halfway	upper	ha4	2004	18.1	11U	450707E	5595218N	883	50	18	21	84%	0.83	1.1%	rp
Halfway	upper	ha5	2004	20.3	11U	452775E	5594894N	1000	50	20	23	87%	0.62	2.1%	rp
Halfway	upper	ha6	2004	23.5	11U	455448E	5593770N	817	50	16	21	78%	0.67	2.1%	rp
Jordan		jr1	2004	2.2	11U	413068E	5652169N	1173	40	29	41	71%	1.20	0.1%	rp
Jordan		jr2	2004	3.8	11U	412891E	5653234N	867	40	22	31	70%	1.09	0.4%	rp
Jordan		jr3	2004	4.7	11U	412236E	5653639N	1055	28	38	42	89%	1.20	1.4%	rp
Kuskanax		ku1	2004	0.9	11U	442148E	5566947N	975	38	26	65	39%	0.84	0.3%	rp
Kuskanax		ku2	2004	2.6	11U	443467E	5567929N	862	47	18	26	71%	0.74	0.1%	rp
Kuskanax		ku3	2004	8.3	11U	447881E	5570488N	1133	50	23	28	81%	1.12	1.0%	rp
Macdld.	lower	mc1	2004	1.2	11U	443544E	5554680N	930	93	10	14	74%	0.37	1.7%	rp
Macdld.	lower	mc2	2004	5.6	11U	446737E	5552044N	371	53	7	13	54%	0.50	5.8%	srp
Macdld.	upper	mc3	2004	8.7	11U	449210E	5550243N	661	115	6	11	52%	0.50	5.6%	srp
Macdld.	upper	mc4	2004	10.2	11U	450866E	5550407N	375	77	5	9	54%	0.31	4.1%	srp
Taite		ta1	2004	0.7	11U	422333E	5516487N	353	53	7	11	61%	0.64	5.4%	srp
Taite		ta2	2004	1.3	11U	422953E	5516778N	291	46	6	9	70%	0.39	5.4%	srp
Taite		ta3	2004	1.9	11U	423364E	5517117N	288	48	6	9	71%	0.56	8.1%	srp

[1] RP: riffle-pool morphology; SRP: step-riffle-pool morphology

Appendix 1b. Description of habitat characteristics for 39 juvenile sampling sites in 10 tributary reaches of the Arrow Lakes Reservoir in 2005.

Stream	Reach	Site	Year	Habitat type(s)[1]					Substrate compos. (%)				D90 (cm)	D50 (cm)	Fish cover			
				Ri	Ru	P	C	SC	Bd	Cb	Gr	Fi			Turbul e-nce (% site area)	Over-head (% site area)	Under-cut (% bank)	LWD (m^2/ linear m)
Caribou	lower	car1	2005		1		1		35	50	10	5	70	22	30	0	0	0.00
Caribou	lower	car2	2005	1	1				30	50	10	10	50	22	20	0	0	0.11
Caribou	upper	car3	2005	1					40	45	10	15	65	30	20	5	10	0.05
Caribou	upper	car3.5	2005	1	1			1	70	25	5	0	130	60	50	10	10	0.06
Caribou	upper	car4	2005	1	1	2	1		40	35	10	15	80	32	20	5	10	0.18
Caribou	upper	car4.5	2005	2		3			30	45	10	15	100	30	20	5	0	0.00
Greely		gr1	2005	4	2		1	2	5	55	30	10	30	12	35	20	10	0.00
Greely		gr2	2005	2	1		2		20	60	15	5	50	20	40	5	10	0.03
Halfway	lower	ha1	2005		2			2	15	75	10	0	40	15	30	0	10	0.00
Halfway	lower	ha2	2005	1				1	30	58	13	0	57	21	20	0	10	0.03
Halfway	lower	ha4	2005	2	1				25	43	43	5	90	25	40	0	3	0.00
Halfway	lower	ha12	2005	1	1				50	45	5	0	90	30	35	0	0	0.00
Halfway	upper	ha6	2005	1	1				25	55	15	5	50	22	30	5	0	0.13
Halfway	upper	ha7	2005	1					25	65	10	0	60	19	40	5	0	0.00
Halfway	upper	ha16	2005	1	1				45	45	10	0	100	50	25	0	5	0.00
Halfway	upper	ha17A	2005	1					30	50	20	0	70	30	0	10	0	0.00
Halfway	upper	ha17B	2005	1	1				20	55	25	0	50	20	20	10	10	0.18
Halfway	upper	ha18	2005	2	2				5	50	40	5	25	12	20	15	0	0.01
Illecill.	lower	il2.0sc	2005					1	15	35	10	40	33	7	0	0	0	0.00
Illecill.	lower	il2.1ru	2005		1				25	25	15	35	60	10	15	0	0	0.00
Illecill.	lower	il2.4ri	2005	1					25	50	23	2	80	18	35	0	0	0.11
Illecill.	upper	il8.8ru	2005		1				15	45	20	20	28	12	30	0	0	1.86
Illecill.	upper	il9.3sc	2005					2	5	30	48	18	13	5	8	0	0	0
Illecill.	upper	il9.9ri	2005	1					30	50	20	0	57	18	30	0	0	0.13
Illecill.	upper	il23.7ri	2005	1					20	55	20	5	30	22	55	2	0	0.35
Illecill.	upper	il23.7sc	2005					1	3	32	35	30	25	13	0	2	5	0.09
Illecill.	upper	il23.8ru	2005		1				25	55	15	5	33	22	15	2	0	0.30
Illecill.	upper	il27.1ri	2005	1					30	50	20	0	120	20	50	0	0	0.10
Illecill.	upper	il27.1ru	2005		1				35	35	15	15	100	15	20	0	0	0.00
Illecill.	upper	il27.3sc	2005					1	10	50	20	20	55	12	10	0	0	0.00
Macdld.	lower	mc1	2005	2	2	1	1		5	50	35	10	22	8	20	5	5	0.33
Macdld.	lower	mc2	2005	1	2		2		40	30	15	15	80	25	50	15	40	0.30
Macdld.	upper	mc3	2005	1		1	2		40	40	10	10	55	25	50	35	10	0.21
Macdld.	upper	mc3.5	2005	2	1	1	2		25	45	20	10	55	22	30	12	10	0.35
Macdld.	upper	mc4	2005	2	2		1		10	25	50	15	35	15	30	15	5	0.26
Macdld.	upper	mc4.5	2005						15	55	20	10	30	10	40	10	0	0.25
Tangier	lower	tan1	2005	1	1													
Tangier	upper	tan2	2005	1		1			60	30	10	0	140	30	40	2	0	0.09
Tangier	upper	tan3	2005	1	1				35	25	35	5	75	10	20	2	0	0.48

[1] Refers to the number of habitat units for each habitat type (riffle, run, pool, cascade, side-channel) occurring in a site.

Appendix 1b (continued)

Stream	Reach	Site	Year	Distance from reservoir (km)	UTM coordinates Zone	Northing	Easting	Site area (m²)	Site length (m)	Wetted width (m)	Channel width (m)	Ratio wetted to channel width	Thalweg depth (m)	Channel slope (%)	Channel type[1]
Caribou	lower	car1	2005	0.6	11.0	437283.0	5536847.0	745	52	14	21	68%	0.59	2.4%	rp
Caribou	lower	car2	2005	3.2	11.0	439500.0	5537086.0	474	29	16	25	66%	0.67	1.0%	rp
Caribou	upper	car3	2005	4.0	11.0	440316.0	5537602.0	1640	120	14	20	70%	0.68	2.0%	rp
Caribou	upper	car3.5	2005	4.2	11U	440442E	5537711N	1467	100	15	20	75%	0.75	3.3%	rp
Caribou	upper	car4	2005	4.6	11.0	440820.0	5538126.0	510	50	10	15	68%	0.83	3.6%	rp
Caribou	upper	car4.5	2005	4.4	11U	440820E	5538126N	717	50	14	21	68%	0.65	2.7%	rp
Greely		gr1	2005	11.0	11.0	425457.0	5652262.0	770	105	7	16	46%	0.39	2.9%	rp
Greely		gr2	2005	10.9	11.0	425533.0	5652045.0	773	92	8	12	72%	0.40	4.3%	srp
Halfway	lower	ha1	2005	3.3	11.0	438075.0	5591126.0	510	24	22	150	14%	0.83	1.2%	rp
Halfway	lower	ha2	2005	4.9	11.0	438498.0	5592513.0	748	40	19	150	13%	0.61	2.2%	rp
Halfway	lower	ha4	2005	8.4	11.0	441470.0	5594196.0	1369	74	19	25	76%	0.79	1.2%	rp
Halfway	lower	ha12	2005	9.5	11U	443324E	5594671N	671	53	13	23	55%	0.71	1.2%	rp
Halfway	upper	ha6	2005	18.1	11.0	450707.0	5595218.0	1320	88	15	21	71%	0.83	0.6%	rp
Halfway	upper	ha7	2005	20.3	11.0	452775.0	5594894.0	1770	90	20	24	82%	0.62	2.1%	rp
Halfway	upper	ha16	2005	27.1	11U	456101E	5590794N	491	46	11	19	56%	0.66	2.4%	rp
Halfway	upper	ha17A	2005	28.2	11U	456404E	5589900N	493	40	12	20	62%	0.48	2.2%	rp
Halfway	upper	ha17B	2005	28.3	11U	456432E	5589884N	836	66	13	29	44%	0.58	1.1%	rp
Halfway	upper	ha18	2005	28.6	11U	456536E	5589532N	1047	125	8	34	25%	0.54	0.8%	rp
Illecill.	lower	il2.0sc	2005	1.7	11U	417763E	5649097N	728	59	12	95	13%	0.37	0.2%	rp
Illecill.	lower	il2.1ru	2005	1.8	11U	417865E	5649043N	1378	53	26	92	28%	1.48	0.2%	rp
Illecill.	lower	il2.4ri	2005	2.2	11U	418174E	5649050N	1439	34	42	115	37%	0.67	1.6%	rp
Illecill.	upper	il8.8ru	2005	9.3	11U	423945E	5651930N	1716	44	39	48	81%	1.11	0.1%	rp
Illecill.	upper	il9.3sc	2005	9.7	11U	424323E	5652173N	105	26	4	95	4%	0.27	0.1%	rp
Illecill.	upper	il9.9ri	2005	10.3	11U	424931E	5652321N	1240	40	31	57	54%	0.90	0.7%	rp
Illecill.	upper	il23.7ri	2005	28.4	11U	436112E	5660821N	1333	43	31	112	28%	0.96	0.4%	rp
Illecill.	upper	il23.7sc	2005	28.3	11U	436099E	5660778N	1100	66	17	100	17%	0.36	0.2%	rp
Illecill.	upper	il23.8ru	2005	28.6	11U	436214E	5660869N	1081	47	23	146	16%	1.24	0.1%	rp
Illecill.	upper	il27.1ri	2005	32.8	11U	437988E	5664106N	667	29	23	27	85%	1.00	0.7%	rp
Illecill.	upper	il27.1ru	2005	32.9	11U	437995E	5664174N	938	42	22	35	64%	1.40	0.2%	rp
Illecill.	upper	il27.3sc	2005	33.1	11U	438157E	5664314N	567	34	17	51	33%	0.73	0.5%	rp
Macdld.	lower	mc1	2005	1.2	11	443544	5554680	569	63	9	14	67%	0.37	1.7%	rp
Macdld.	lower	mc2	2005	5.6	11	446737	5552044	511	73	7	13	54%	0.50	5.8%	srp
Macdld.	upper	mc3	2005	8.7	11	449210	5550243	575	100	6	11	52%	0.50	5.6%	srp
Macdld.	upper	mc3.5	2005	8.9				745	98	8	12	63%	0.46	7.1%	srp
Macdld.	upper	mc4	2005	10.2	11	450866	5550407	507	100	5	9	56%	0.31	4.1%	srp
Macdld.	upper	mc4.5	2005	10.4	11U	450972E	5550186N	520	100	5	9	58%	0.43	4.2%	srp
Tangier	lower	tan1	2005	37.2				1320	60	22	26	85%			rp
Tangier	upper	tan2	2005	42.4	11U	441708E	5670549N	1128	47	24	27	89%	0.84	1.1%	srp
Tangier	upper	tan3	2005	45.6	11U	443178E	5673085N	1368	48	29	31	92%	0.60	1.2%	rp

[1] RP: riffle-pool morphology; SRP: step-riffle-pool morphology

Appendix 1c. Description of habitat characteristics for 41 juvenile sampling sites in 10 tributary reaches of the Arrow Lakes Reservoir in 2006.

Stream	Reach	Site	Year	Habitat type(s)[1] Ri	Ru	P	C	SC	Substrate compos. (%) Bd	Cb	Gr	Fi	D90 (cm)	D50 (cm)	Fish cover: Turb-ulence (% site area)	Over-head (% site area)	Under-cut (% bank)	LWD (m^2/ linear m)
Incom	lower	in1.4ru	2006		1				30	35	0	35	50	10	0	0	0	0.18
Incom	lower	in1.8b	2006					1	0	0	0	100	1	0	0	0	0	0.04
Incom	lower	in1.9ri	2006	1					0	5	60	35	10	6	30	0	0	0.84
Incom	upper	in5.94b	2006					1	0	5	35	60	12	4				0.20
Incom	upper	in5.95ru	2006		1				20	0	20	60	100	20	10			0.00
Incom	upper	in5.9ri	2006	1					0	30	40	30	25	10	10			0.00
Incom	upper	in9.04b	2006					1	0	5	35	60	8	2				0.16
Incom	upper	in9.53ru	2006	1	1				0	10	30	60	10	2	20			1.47
Incom	upper	in9.57ri	2006	1					0	15	55	30	13	6	0		0	0.06
Incom	upper	in13.9b	2006					1	0	10	20	70	8	0	10	10		0.78
Incom	upper	in14.0ru	2006		1				0	30	50	15	10	5	15	0		0.28
Incom	upper	in14.1ri	2006	1					0	15	70	15	6	2	35	2		0.68
Incom	upper	in17.4b	2006					1	0	45	45	10	15	7	25			0.02
Incom	upper	in17.4ri	2006	1					0	70	30	0	18	9	60			0.00
Incom	upper	in17.5ru	2006		1				0	45	40	15	19	6	15			0.24
Incom	upper	in19.1b	2006					1	0	5	65	30	6	2	5	0		0.76
Incom	upper	in19.1ru	2006		1				0	35	50	15	18	6	0	0		0.56
Incom	upper	in22.3ru	2006		1				5	50	25	20	28	13	30			0.00
Incom	upper	in23.8b2	2006					1	0	0	60	40	6	3				0.30
Incom	upper	in23.9b	2006					1	10	30	40	20	25	13				0.40
Incom	upper	in23.9ri	2006	1					0	50	40	10	30	12	10			0.07
Incom	upper	in24.0ru	2006		1				0	30	40	30	20	12	10			0.50
Incom	upper	in27.3ri	2006	1					20	60	10	10	60	25	20			0.00
Incom	upper	in27.4ru	2006		1				20	45	15	20	60	25	10			0.00
Lexing		lex1	2006						40	30	10	20	120	30	30	50	5	0.17
Menhin		men1	2006						40	30	25	5	50	25	10	15	10	0.04
Sable		sa1	2006	1	2			1	0	10	30	60	8	6	5	5	0	0.27
Sable		sa2	2006	2	1			1	0	20	40	40	15	7	20	0	0	0.31
Boyd		boy1	2006						15	55	15	15	30	9	10	5		0.30
Boyd		boy2	2006						0	15	50	35	8	3	10			1.00
Pool		po1	2006	1					40	40	10	10	120	25	40	4	0	0.00
Pool		po2	2006				1		50	30	10	10	150	35	60	5	0	0.00
Kelly		ke1	2006						0	5	40	55	7	0	20			0.00
Kelly		ke2	2006						0	0	30	70	3	0	0	5	25	0.90
Illecillew	upper	il17.9ri	2006	1					60	20	20	0	40	15	5	0	0	0.04
Illecillew	upper	il18.1ru	2006		1				40	30	20	10	60	25	5	0	0	0.02
Illecillew	upper	il30.5ri	2006	1					20	50	25	5	55	16	40			0.11
Illecillew	upper	il30.6b	2006					1	25	35	25	15	50	16				0.00
Illecillew	upper	il30.9ru	2006		1				35	35	10	20	50	22	5			0.64
Albert		al1	2006						50	40	8	2	70	25	40			0.00
Albert		al2	2006						20	55	25	0	45	20	25	5		0.16

Appendix 1c (continued)

Stream	Reach	Site	Year	UTM coordinates Zone	Northing	Easting	Site area (m^2)	Site length (m)	Wet-ted width (m)	Chan-nel width (m)	Ratio wetted to chan-nel width	Thal-weg depth (m)	Chan-nel slope (%)	Chan-nel type[1]
Incom	lower	in1.4ru	2006	11 U	450844.0	5622784.0	2964	57	52	63	83%	0.69		rp
Incom	lower	in1.8b	2006	11 U	451061.0	5623126.0	656	48	14	58	24%	0.71		rp
Incom	lower	in1.9ri	2006	11 U	451081.0	5623185.0	1900	50	38	43	88%	0.27		rp
Incom	upper	in5.9ri	2006	11 U	453828.0	5626241.0	2217	50	44	75	59%	0.21	0.5%	rp
Incom	upper	in5.94b	2006	11 U	453843.0	5626178.0	660	60	11	100	11%	0.51	0.1%	rp
Incom	upper	in5.95ru	2006	11 U	453743.0	5626230.0	3520	64	55	75	73%	1.88	0.1%	rp
Incom	upper	in9.04b	2006	11 U	455316.0	5629020.0	500	50	10	105	10%	0.78	0.1%	rp
Incom	upper	in9.57ri	2006	11 U	455789.0	5629220.0	1988	53	38	83	45%	0.30	0.5%	rp
Incom	upper	in9.53ru	2006	11 U	455874.0	5629313.0	2720	49	56	72	88%	0.80	0.2%	rp
Incom	upper	in13.9b	2006	11 U	456138.0	5634261.0	460	58	8	70	12%	0.83		rp
Incom	upper	in14.1ri	2006	11 U	456082.0	5634519.0	2405	37	65	70	93%	0.54		rp
Incom	upper	in14.0ru	2006	11 U	456119.0	5634410.0	3024	72	42	68	62%	0.88		rp
Incom	upper	in17.4b	2006	11 U	458011.0	5637260.0	354	55	7	300	10%	0.31		rp
Incom	upper	in17.4ri	2006	11 U	457967.0	5637330.0	1107	40	28	300	9%	0.44		rp
Incom	upper	in17.5ru	2006	11 U	458089.0	5637404.0	2125	63	34	120	28%	0.93		rp
Incom	upper	in19.1b	2006	11 U	458801.0	5638767.0	105	42	3	150	2%	0.20		rp
Incom	upper	in19.1ru	2006	11 U	458756.0	5638861.0	1458	54	27	150	18%	0.82		rp
Incom	upper	in22.3ru	2006	11 U	458415.0	5642409.0	966	46	21	180	12%	0.82	0.1%	rp
Incom	upper	in23.9b	2006	11 U	457905.0	5644316.0	130	50	3	150	2%	0.37	0.4%	rp
Incom	upper	in23.8b2	2006	11 U	457990.0	5644291.0	367	50	7	180	4%	0.43		rp
Incom	upper	in23.9ri	2006	11 U	457953.0	5644384.0	915	30	31	54	56%	0.33	0.7%	rp
Incom	upper	in24.0ru	2006	11 U	457918.0	5644455.0	1400	50	28	35	80%	0.99	0.1%	rp
Incom	upper	in27.3ri	2006	11 U	458385.0	5647755.0	1024	45	23	42	55%	0.69	1.0%	rp
Incom	upper	in27.4ru	2006	11 U	458468.0	5647816.0	968	43	23	33	68%	0.99	0.7%	rp
Lexing		lex1	2006	11 U	456162	5632492	189	52	4	12	30%	0.31		sp
Menhin		men1	2006	11 U	454925	5628965	413	55	8	18	42%	0.33		rp
Sable		sa1	2006	11 U	455424	5632710	809	71	11	18	63%	0.55		rp
Sable		sa2	2006	11 U	455436	5632991	663	59	11	19	60%	0.43		rp
Boyd		boy1	2006	11 U	459143	5639126	550	50	11	27	41%	0.55	2.5%	srp
Boyd		boy2	2006	11 U	458937	5639073	650	50	13	43	30%	0.57	1.5%	rp
Pool		po1	2006	11 U	454549	5626722	517	50	10	15	69%	0.57	3.5%	srp
Pool		po2	2006	11 U	454877	5626835	360	47	8	12	64%	0.59	4.5%	sp
Kelly		ke1	2006	11 U	459747	5640750	512	64	8	16	50%	0.95	2.3%	rp
Kelly		ke2	2006	11 U	459323	5640485	617	50	12	20	62%	0.93	1.0%	rp
Illecillew	upper	il17.9ri	2006	11 U	432563	5654592	1383	50	28	50	55%	0.59		rp
Illecillew	upper	il18.1ru	2006	11 U	432668	5654740	1200	60	20	30	67%	0.93		rp
Illecillew	upper	il30.5ri	2006	11 U	440843	5665922	720	36	20	31	65%	0.46	1.0%	rp
Illecillew	upper	il30.6b	2006	11 U	440949	5665954	278	58	5	30	16%	0.16	0.5%	rp
Illecillew	upper	il30.9ru	2006	11 U	441116	5665972	840	56	15	43	35%	1.00	0.5%	rp
Albert		al1	2006	11 U	439661	5664559	473	53	9	16	56%	0.52		srp
Albert		al2	2006	11 U	439611	5665271	693	50	14	32	44%	0.48		rp

[1] RP: riffle-pool morphology; SRP: step-riffle-pool morphology

Appendix 2a. Summary of scale data collected for bull trout in Arrow Lakes Reservoir tributaries in 2004 and 2005. For older fish, lengths at earlier ages were estimated using a back-calculation method based on the Fraser-Lee equation (Duncan 1980).

Sample	Stream	Fork length (mm)	Age	Scale radius	Annulus				Est. Length/annulus			
					1	2	3	4	1	2	3	4
1	Caribou	94	1+	14	6				49			
2	Caribou	100	1+	15	8				61			
3	Caribou	100	1+	18	7				49			
4	Caribou	105	1+	13	7				64			
5	Caribou	108	1+	18	8				57			
6	Caribou	110	1+	15	8				66			
7	Caribou	112	1+	14	9				78			
8	Caribou	116	1+	19	9				63			
9	Caribou	117	1+	16	7				60			
10	Caribou	117	1+	16	7				60			
11	Caribou	120	1+	15	7				64			
12	Caribou	125	1+	18	9				70			
13	Caribou	157	2+	26	10	18			70	114		
14	Caribou	165	2+	23	10	17			81	126		
15	Greeley	105	1+	16	7				55			
16	Greeley	107	1+	14	7				61			
17	Greeley	135	1+	19	7				60			
18	Greeley	175	2+	25	6	13			54	99		
19	Halfway	45	0+	5								
20	Halfway	45	0+	5								
21	Halfway	85	1+	13	8				58			
22	Halfway	85	1+	16	9				55			
23	Halfway	90	1+	17	8				51			
24	Halfway	90	1+	13	6				50			
25	Halfway	95	1+	15	7				53			
26	Halfway	95	1+	13	7				58			
27	Halfway	97	1+	15	7				54			
28	Halfway	100	1+	17	8				55			
29	Halfway	103	1+	16	7				54			
30	Halfway	110	1+	14	6				56			
31	Halfway	125	2+	19	8	15			62	102		
32	Halfway	125	2+	21	8	17			57	104		
33	Halfway	125	2+	23	8	16			54	92		
34	Halfway	127	2+	19	6	13			51	92		
35	Halfway	128	2+	20	7	15			55	100		
36	Halfway	130	2+	19	6	13			52	94		
37	Halfway	130	2+	19	6	13			52	94		
38	Halfway	137	2+	22	7	17			54	109		
39	Halfway	140	2+	23	10	20			70	124		
40	Halfway	140	2+	18	7	12			64	99		
41	Halfway	145	2+	22	na	16				110		
42	Halfway	145	2+	21	6	14			53	102		
43	Halfway	168	2+	25	10	19			77	131		
44	Halfway	185	3+	31	8	17	24		60	109	147	
45	Halfway	195	4+	32	5	14	21	28	44	94	133	173

Appendix 2a (continued)

Sample	Stream	Fork length (mm)	Age	Scale radius	Annulus				Est. Length/annulus			
					1	2	3	4	1	2	3	4
46	Halfway	230	4+	36	7	14	19	26	58	99	129	171
47	Halfway	130	R	16								
48	Halfway Trib.	204	4+	28	6	11	17	24	56	90	130	
49	MacDonald	74	1+	10	6				51			
50	MacDonald	84	1+	12	5				44			
51	MacDonald	84	1+	12	7				56			
52	MacDonald	85	1+	17	6				40			
53	MacDonald	85	1+	14	6				46			
54	MacDonald	95	1+	14	9				67			
55	MacDonald	96	1+	15	8				59			
56	MacDonald	97	1+	14	8				62			
57	MacDonald	100	1+	16	6				47			
58	MacDonald	102	1+	14	7				59			
59	MacDonald	105	1+	15	7				57			
60	MacDonald	110	1+	18	9				63			
61	MacDonald	110	1+	13	9				81			
62	MacDonald	118	2+	14	7	12			67	103		
63	MacDonald	122	2+	22	9	16			59	93		
64	MacDonald	124	2+	17	8	14			67	105		
65	MacDonald	125	2+	21	9	15			63	94		
66	MacDonald	125	2+	21	7	15			52	94		
67	MacDonald	135	2+	18		13				102		
68	MacDonald	135	2+	22	7	15			54	97		
69	MacDonald	140	2+	19	7	14			62	107		
70	MacDonald	143	2+	21	6	15			52	107		
71	MacDonald	145	2+	19	8	15			70	118		
72	MacDonald	145	2+	18	8	13			73	109		
73	MacDonald	145	2+	22	7	15			57	104		
74	MacDonald	150	2+	22		13				95		
75	MacDonald	152	2+	21	8	14			68	107		
76	MacDonald	154	2+	20	7	13			64	106		
77	MacDonald	154	2+	25	8	16			60	104		
78	MacDonald	155	2+	22	7	16			60	117		
79	MacDonald	168	2+	22	8	15			71	120		
80	MacDonald	180	2+	25	8	16			68	121		
81	MacDonald	170	3+	24		12	20			93	144	
82	MacDonald	180	3+	25	6	11	19		55	88	141	
83	St. Leon	99	1+	15	6				49			
84	St. Leon	118	1+	15	9				77			
85	West Twin	50	0+	7								
86	West Twin	102	1+	17	8				56			
87	West Twin	110	1+	16	8				63			
88	West Twin	110	1+	19	7				51			
89	West Twin	154	2+	21	6	14			55	108		
90	West Twin	178	2+	28	8	19			62	126		

Appendix 2b. Summary of scale data collected for rainbow trout in Arrow Lakes Reservoir tributaries in 2004 and 2005. For older fish, lengths at earlier ages were estimated using a back-calculation method based on the Fraser-Lee equation (Duncan 1980).

		Fork length		Scale	Annulus				Est. Length/annulus			
Sample	Stream	(mm)	Age	radius	1	2	3	4	1	2	3	4
1	Caribou	130	2+	27	11	21			74	109		
2	Caribou	176	2+	33	8	22			70	129		
3	Caribou	170	3+	29		12	22			91	138	
4	Halfway	84	1+	14	8				63			
5	Halfway	98	1+	21	8				59			
6	Halfway	115	1+	26	6	19			54	94	??	
7	Halfway	118	2+	26		19				96		
8	Halfway	120	2+	17		12				95		
9	Halfway	125	2+	30	9	23			62	104		
10	Halfway	129	2+	27		21				108		
11	Halfway	129	2+	28	7	20			59	102		
12	Halfway	165	2+	39	9	25			65	119		
13	Halfway	140	3+	29		14	22			86	115	
14	Halfway	140	3+	29	11	19	25		75	104	126	
15	Halfway	146	3+	30		13	23			83	120	
16	Halfway	149	3+	28	6	15	23		60	96	129	
17	Halfway	150	3+	38	11	21	32		69	99	132	
18	Halfway	158	3+	31	11	20	27		79	115	142	
19	Halfway	160	3+	31	7	17	26		64	104	140	
20	Halfway	170	3+	32	6	16	26		61	103	145	
21	Halfway	172	4+	43	6	16	27	36	55	86	121	150
22	Halfway	175	4+	42	5	18	31	38	52	95	138	162
23	Halfway	180	4+	42		16	24	33		91	118	149
24	Halfway	185	4+	46	8	20	29	38	62	101	130	159
25	Halfway	185	4+	26	6	15	24	31	70	122	174	214
26	Halfway	195	4+	37		16	23	32		105	135	173
27	Halfway	233	regenerated									
28	MacDonald	91	1+	20	8				58			
29	MacDonald	132	2+	29	8	22			62	109		
30	MacDonald	135	2+	26		15				93		
31	MacDonald	155	2+	42	10	39			64	146		
32	MacDonald	155	2+	36	10	24			69	115		
33	MacDonald	175	2+	43	8	23			62	110		
34	MacDonald	234	4+	40	7	21	30	37	70	140	184	219
35	St. Leon	93	1+	18	8				61			
36	St. Leon	128	1+	27	11				73			
37	Tait	42	0+	7								
38	Tait	85	1+	17	7				56			
39	Tait	85	1+	19	9				59			
40	Tait	128	2+	30	9	24			63	110		
41	Tait	185	3+	32	6	20	29		64	129	171	
42	Tait	171	4+	40		20	29	37		103	134	161
43	Tait	130	regenerated									
44	Tait	192	regenerated									

Appendix 3a. Mark-recapture data collected to estimate snorkeling efficiency for bull trout at individual Arrow Lakes Reservoir tributary sites in 2004 and 2005. Data are shown for two fork length classes (<110 mm and >100 mm) and for two capture methods for fish marking (snorkeler-capture or electrofishing-capture). Where possible, we combined data from sites with few marked fish (< 10) with data from a nearby marking site within the same stream to provide a single mark-recapture estimate.

Site	Year	Marks <110 mm	Marks >110 mm	Total marks	Total captures	Recaptures <110 mm	Recaptures >110 mm	Recaptures total	Recaptures d/s	Recaptures u/s	*p*	Asin sqrt *p*	Pop estimate
Snorkeler-captured marks													
car3*	2004			4	11			3	0	0			
car4*	2004			11	26			7	0	0			
car3+4	2004			15	37			10	0	0	0.67	54.7	55
ha6*	2004												
ha7*	2004												
ha4&5	2004			10	27			7	0	0	0.70	56.8	39
mc3	2004			7	22			2	0	0	0.29	32.3	61
car3	2005	7	7	14	36	4	3	7	0	0	0.50	45.0	69
car3.5	2005	8	4	12	32	3	4	7	0	0	0.58	49.8	54
car4	2005	6	8	14	32	3	5	8	0	0	0.57	49.1	55
car4.5	2005	7	5	12	17	3	1	4	0	0	0.33	35.3	47
ha17*	2005	7	6	13	14	2	3	5	0	0			
ha18*	2005	2	6	8	18	1	4	5	0	0			
ha17+18	2005	9	12	21	32	3	7	10	0	0	0.48	43.6	66
ha7	2005	5	5	10	11	1	2	3	0	0	0.30	33.2	33
mc3*	2005	0	8	8	20		2	2	0	0			
mc4*	2005	3	8	11	26	2	4	6	0	0			
mc3+4	2005	3	16	19	46	2	6	8	0	0	0.42	40.5	104
mc4.5	2005	3	7	10	17	1	4	5	0	0	0.50	45.0	33
Electrofishing-captured marks													
gr1	2004			10	34			1	0	0	0.10	18.4	193
gr2	2004			12	31			4	0	0	0.33	35.3	83
ha7	2004			7	17			3	0	0	0.43	40.9	36
mc4b	2004			10	18			1	0	0	0.10	18.4	105
gr2	2005	5	6	11	42	2	2	4	0	0	0.36	37.1	103
mc3.5	2005	5	12	17	12	1	2	3	0	0	0.18	24.8	59

* denotes site data combined with data from another site in same reach because of insufficient sample size in one or both sites

Appendix 3b. Mark-recapture data collected to estimate snorkeling efficiency for rainbow trout at individual Arrow Lakes Reservoir tributary sites in 2004 and 2005. Data are shown for three fork length classes (<100 mm, 100-170 mm, and >170 mm) and for two capture methods for fish marking (snorkeler-capture or electrofishing-capture). Where possible, we combined data from sites with few marked fish (< 10) with data from a nearby marking site within the same stream to provide a single mark-recapture estimate.

		Marks				Total	Recaptures							Asin	Pop
Site	Year	<100	100-170	>170	Total	captures	<100	100-170	>170	total	d/s	u/s	*p*	sqrt *p*	estimate
Snorkeler-captured marks															
car3*	2004				7	16				6	1	0			
car4*	2004				8	18				3	1	0			
car3+4	2004				15	34				9	2	0	0.60	50.8	56
car3*	2005	0	1	4	5	24	1	0	5	6	0	0			
car3.5*	2005	0	6	2	8	42	0	6	2	8	0	0			
car4*	2005	0	1	6	7	25	0	4	3	7	1	0			
car3,3.5,4	2005	0	8	12	20	91	1	10	10	21	1	0	1.05	90.0	88
ha1	2004				18	58				11	0	1	0.61	51.4	93
ha3	2004				31	98				17	0	0	0.55	47.8	176
ha6*	2004														
ha7*	2004														
ha6&7	2004				19	56				15	1	2	0.79	62.7	71
mc2*	2004				14	38				12	1	0			
mc3*	2004				4	11				2	0	0			
mc2+3	2004				18	49				14	1	0	0.78	61.9	63
ta1	2004				15	54				10	0	0	0.67	54.7	80
ta2*	2004														
ta3*	2004														
ta2&3	2004				14	52				8	0	0	0.57	49.1	88
car4.5	2005	4	7	4	15	32	2	4	4	10	0	1	0.67	54.7	48
ha17*	2005	0	4	4	8	14	0	3	4	7	0	0			
ha7*	2005	2	14	5	21	55	0	12	4	16	0	0			
ha17+5	2005	2	18	9	29	69	0	15	8	23	0	0	0.79	62.9	88
mc3*	2005	2	3	5	10	18	0	2	4	6	0	0			
mc4*	2005	0	4	1	5	7	0	3	0	3	0	0			
mc4.5*	2005	0	4	0	4	9	0	3	0	3	0	0			
mc3,4,4.5	2005	2	11	6	19	34	0	8	4	12	0	0	0.63	52.6	54
Electrofishing-captured marks															
ha4	2004	8			14	118				4	0	3	0.29	32.3	357
ha2*	2004				9	48				5	0	0			
ha7*	2004				4	22				2	0	0			
ha11+5b	2004				13	70				7	0	0	0.54	47.2	124
mc3.5*	2005	0	1	0	1	4	0	0	0	0	0	0			
mc4*	2004				4	11				2	0	0			
mc3.5+4	04+05				4	11				2	0	0	0.50	45.0	20

* denotes site data combined with data from another site in same reach because of insufficient sample size in one or both sites

Appendix 4a. Summary of estimated bull trout parr densities by age class and by site in Arrow Lakes Reservoir tributary reaches in 2004. Estimates are based on snorkeler counts adjusted to account for snorkeling efficiency. 95% confidence intervals for the density estimates (shown for fish/100m^2 only) reflect error in the estimation of snorkeling efficiency.

			Sampling	Fish/100m									Fish/100m^2		
			temp	Age 1+			Age 2+			Age 3+			1+	2+	3+
Stream	Reach	Site	(°C)	est.	LCI	UCI	est.	LCI	UCI	est.	LCI	UCI	est.	est.	est.
Burton		bu1	11.9	4.9	3.1	11.7	0.0	-	-	0.0	-	-	0.5	0.0	0.0
Burton		bu2	11.9	0.0	-	-	4.0	2.6	9.6	0.0	-	-	0.0	0.3	0.0
Caribou	lower	car1	12.6	0.0	-	-	0.0	-	-	0.0	-	-	0.0	0.0	0.0
Caribou	lower	car2	12.6	0.0	-	-	3.7	2.3	8.7	0.0	-	-	0.0	0.2	0.0
Caribou	upper	car3	12.0	44.9	28.5	106	4.5	2.8	10.6	0.0	-	-	2.8	0.3	0.0
Caribou	upper	car4	11.7	141	89.3	334	12.2	7.8	29.0	0.0	-	-	11.1	1.0	0.0
Greeley		gr1	8.3	57.7	36.6	137	3.8	2.4	9.1	1.9	1.2	4.6	6.4	0.4	0.2
Greeley		gr2	8.3	71.5	45.3	169	4.9	3.1	11.7	0.0	-	-	8.6	0.6	0.0
Halfway	lower	ha1	12.0	0.0	-	-	0.0	-	-	0.0	-	-	0.0	0.0	0.0
Halfway	lower	ha11	12.0	0.0	-	-	0.0	-	-	0.0	-	-	0.0	0.0	0.0
Halfway	lower	ha10	11.8	0.0	-	-	4.0	2.6	9.6	0.0	-	-	0.0	0.2	0.0
Halfway	lower	ha2	11.8	0.0	-	-	0.0	-	-	0.0	-	-	0.0	0.0	0.0
Halfway	upper	ha3	11.3	0.0	-	-	4.0	2.6	9.6	4.0	2.6	9.6	0.0	0.2	0.2
Halfway	upper	ha4	8.3	20.2	12.8	47.9	24.3	15.4	57.4	0.0	-	-	1.1	1.4	0.0
Halfway	upper	ha5	8.6	46.5	29.5	110	20.2	12.8	47.9	0.0	-	-	2.3	1.0	0.0
Halfway	upper	ha6	8.6	0.0	-	-	4.0	2.6	9.6	0.0	-	-	0.0	0.2	0.0
Jordan		jr1	8.0	0.0	-	-	0.0	-	-	0.0	-	-	0.0	0.0	0.0
Jordan		jr2	8.0	0.0	-	-	0.0	-	-	0.0	-	-	0.0	0.0	0.0
Jordan		jr3	8.0	21.7	13.7	51.3	0.0	-	-	0.0	-	-	0.6	0.0	0.0
Kuskanax		ku1	12.0	0.0	-	-	0.0	-	-	0.0	-	-	0.0	0.0	0.0
Kuskanax		ku2	12.0	4.3	2.7	10.2	0.0	-	-	0.0	-	-	0.2	0.0	0.0
Kuskanax		ku3	11.0	8.1	5.1	19.1	0.0	-	-	0.0	-	-	0.4	0.0	0.0
Macdld.	lower	mc1	11.7	0.0	-	-	0.0	-	-	0.0	-	-	0.0	0.0	0.0
Macdld.	lower	mc2	11.1	3.8	2.4	9.0	7.6	4.8	18.1	0.0	-	-	0.5	1.1	0.0
Macdld.	upper	mc3	10.4	31.6	20.0	74.9	7.0	4.5	16.7	0.0	-	-	5.5	1.2	0.0
Macdld.	upper	mc4	10.4	36.7	23.3	87.0	10.5	6.7	24.9	0.0	-	-	7.5	2.2	0.0
Taite		ta1	12.0	0.0	-	-	0.0	-	-	0.0	-	-	0.0	0.0	0.0
Taite		ta2	11.5	4.4	2.8	10.4	0.0	-	-	0.0	-	-	0.7	0.0	0.0
Taite		ta3	11.5	0.0	-	-	0.0	-	-	0.0	-	-	0.0	0.0	0.0

Appendix 4b. Summary of estimated bull trout parr densities by age class and by site in Arrow Lakes Reservoir tributary reaches in 2005. Estimates are based on snorkeler counts adjusted to account for snorkeling efficiency. Shaded rows show mean density values for Illecillewaet River sampling locations, which consisted of three sites in three different habitat types (run/pool, riffle, side-channel/braid).

			Sampling	Fish/100m									Fish/100m²		
			temp	Age 1+			Age 2+			Age 3+			1+	2+	3+
Stream	Reach	Site	(°C)	est.	LCI	UCI	est.	LCI	UCI	est.	LCI	UCI	est.	est.	est.
Caribou	lower	car1	11.2	0.0	-	-	0.0	-	-	0.0	-	-	0.0	0.0	0.0
Caribou	lower	car2	11.4	20.9	13.2	49.5	7.0	4.4	16.5	0.0	-	-	1.3	0.4	0.0
Caribou	upper	car3	11.8	37.1	23.5	87.8	20.2	12.8	47.9	3.4	2.1	8.0	2.7	1.5	0.2
Caribou	upper	car3.5	11.3	54.6	34.6	129	10.1	6.4	23.9	0.0	-	-	3.7	0.7	0.0
Caribou	upper	car4	10.5	93.0	58.9	220	36.4	23.1	86.2	0.0	-	-	9.1	3.6	0.0
Caribou	upper	car4.5	10.5	56.6	35.9	134	12.1	7.7	28.7	0.0	-	-	3.9	0.8	0.0
Halfway	lower	ha1	12.0	0.0	-	-	8.5	5.4	20.2	0.0	-	-	0.0	0.4	0.0
Halfway	lower	ha2	11.0	0.0	-	-	0.0	-	-	0.0	-	-	0.0	0.0	0.0
Halfway	lower	ha4	10.5	0.0	-	-	0.0	-	-	0.0	-	-	0.0	0.0	0.0
Halfway	lower	ha12	10.5	0.0	-	-	3.8	2.4	9.0	0.0	-	-	0.0	0.3	0.0
Halfway	upper	ha6	8.2	6.9	4.4	16.3	2.3	1.5	5.4	2.3	1.5	5.4	0.5	0.2	0.2
Halfway	upper	ha7	8.2	11.2	7.1	26.6	13.5	8.5	31.9	0.0	-	-	0.6	0.7	0.0
Halfway	upper	ha16	8.4	17.6	11.1	41.6	17.6	11.1	41.6	4.4	2.8	10.4	1.6	1.6	0.4
Halfway	upper	ha17A	12.5	0.0	-	-	15.2	9.6	35.9	0.0	-	-	0.0	1.2	0.0
Halfway	upper	ha17B	10.0	12.2	7.8	29.0	9.2	5.8	21.8	3.1	1.9	7.3	1.0	0.7	0.2
Halfway	upper	ha18	8.4	9.7	6.1	23.0	12.9	8.2	30.6	4.9	3.1	11.5	1.2	1.5	0.6
Greeley		gr1	9.2	50.0	31.7	119	21.2	13.4	50.1	0.0	-	-	6.8	2.9	0.0
Greeley		gr2	9.3	59.3	37.6	140	28.6	18.1	67.6	2.2	1.4	5.2	7.1	3.4	0.3
Illecillew	lower	il2.0sc	3.2	0.0	-	-	0.0	-	-	0.0	-	-	0.0	0.0	0.0
Illecillew	lower	il2.1ru	3.0	0.0	-	-	0.0	-	-	0.0	-	-	0.0	0.0	0.0
Illecillew	lower	il2.4ri	3.0	0.0	-	-	0.0	-	-	0.0	-	-	0.0	0.0	0.0
Illecillew	lower	il_Revel	weighted mean	0.0	-	-	0.0	-	-	0.0	-	-	0.0	0.0	0.0
Illecillew	upper	il8.8ru	3.2	45.9	29.1	109	13.8	8.7	32.6	9.2	5.8	21.8	1.2	0.4	0.2
Illecillew	upper	il9.3sc	3.2	54.4	34.5	129	15.5	9.9	36.8	0.0	-	-	6.6	1.9	0.0
Illecillew	upper	il9.9ri	3.0	80.8	51.2	191	25.3	16.0	59.8	0.0	-	-	2.6	0.8	0.0
Illecillew	upper	il_Greeley	weighted mean	69.4	44.0	164	21.2	13.4	50.1	5.1	3.3	12.2	1.9	0.6	0.1
Illecillew	upper	il23.7ri	3.2	4.7	3.0	11.1	0.0	-	-	0.0	-	-	0.2	0.0	0.0
Illecillew	upper	il23.7sc	3.2	3.1	1.9	7.3	0.0	-	-	0.0	-	-	0.2	0.0	0.0
Illecillew	upper	il23.8ru	3.2	17.2	10.9	40.7	4.3	2.7	10.2	0.0	-	-	0.7	0.2	0.0
Illecillew	upper	il_Skunk	weighted mean	12.0	7.6	28.4	2.1	1.3	4.9	0.0	-	-	0.4	0.1	0.0
Illecillew	upper	il27.1ri	2.0	34.8	22.1	82.5	13.9	8.8	33.0	0.0	-	-	1.5	0.6	0.0
Illecillew	upper	il27.1ru	2.0	67.4	42.7	160	19.2	12.2	45.6	4.8	3.0	11.4	3.0	0.9	0.2
Illecillew	upper	il27.3sc	2.0	35.7	22.6	84.5	29.7	18.8	70.4	11.9	7.5	28.2	2.1	1.8	0.7
Illecillew	upper	il_Tunnel	weighted mean	61.3	38.9	145	25.6	16.3	60.7	6.0	3.8	14.1	2.7	1.1	0.3
Macdld.	lower	mc1	9.5	0.0	-	-	0.0	-	-	0.0	-	-	0.0	0.0	0.0
Macdld.	lower	mc2	9.6	5.5	3.5	13.1	13.8	8.8	32.8	0.0	-	-	0.8	2.0	0.0
Macdld.	upper	mc3	8	10.1	6.4	23.9	30.3	19.2	71.8	0.0	-	-	1.8	5.3	0.0
Macdld.	upper	mc3.5	9	18.6	11.8	44.0	6.2	3.9	14.7	0.0	-	-	2.4	0.8	0.0
Macdld.	upper	mc4	8.6	18.2	11.5	43.1	32.3	20.5	76.6	2.0	1.3	4.8	3.6	6.4	0.4
Macdld.	upper	mc4.5	8.6	2.0	1.3	4.8	28.3	17.9	67.0	4.0	2.6	9.6	0.4	5.4	0.8
Tangier	lower	tan1	1.3	3.4	2.1	8.0	16.8	10.7	39.9	3.4	2.1	8.0	0.2	0.8	0.2
Tangier	upper	tan2	1.3	0.0	-	-	4.3	2.7	10.2	4.3	2.7	10.2	0.0	0.2	0.2
Tangier	upper	tan3	1.2	4.2	2.7	10.0	8.4	5.3	19.9	0.0	-	-	0.1	0.3	0.0

Appendix 4c. Summary of estimated bull trout parr densities by age class and by site in Arrow Lakes Reservoir tributary reaches in 2006. Estimates are based on snorkeler counts adjusted to account for snorkeling efficiency. Shaded rows show mean density values for Illecillewaet and Incomappleaux sampling locations, which consisted of 2-5 sites representing different habitat types (run/pool, riffle, side-channel/braid).

			Sampling	Fish/100m									Fish/100m^2		
			temp	Age 1+			Age 2+			Age 3+			1+	2+	3+
Stream	Reach	Site	(°C)	est.	LCI	UCI	est.	LCI	UCI	est.	LCI	UCI	est.	est.	est.
Incomap	upper	in27.3ri		4.5	3.2	12.0	0.0	0.0	0.0	0.0	0.0	0.0	0.2	0.0	0.0
Incomap	upper	in27.4ru		79.9	54.4	203	14.1	9.1	34.2	9.4	5.6	20.8	3.6	0.6	0.4
Incomap	upper	in_BattleBrook		45.5	31.1	116	7.7	5.0	18.6	5.1	3.0	11.3	2.0	0.3	0.2
Incomap	upper	in23.9ri		0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Incomap	upper	in22.3ru		8.8	6.4	23.9	4.4	2.4	8.8	0.0	0.0	0.0	0.4	0.2	0.0
Incomap	upper	in24.0ru		16.2	12.8	47.9	4.0	2.4	8.8	0.0	0.0	0.0	0.6	0.1	0.0
Incomap	upper	in23.8b2		12.1	9.6	35.9	8.1	4.7	17.6	4.0	2.8	10.4	1.7	1.1	0.6
Incomap	upper	in23.9b		4.0	3.2	12.0	20.2	11.8	43.9	4.0	2.8	10.4	1.6	7.8	1.6
Incomap	upper	in_McDougal		13.3	10.3	38.6	10.6	6.1	22.9	2.2	1.5	5.6	1.2	2.5	0.6
Incomap	upper	in17.4ri	6.0	10.1	6.4	23.9	0.0	0.0	0.0	0.0	0.0	0.0	0.4	0.0	0.0
Incomap	upper	in17.5ru	6.0	16.2	16.0	59.8	12.9	9.4	35.1	0.0	0.0	0.0	0.5	0.4	0.0
Incomap	upper	in19.1ru	4.5	33.7	28.8	108	3.7	2.4	8.8	0.0	0.0	0.0	1.2	0.1	0.0
Incomap	upper	in17.4b	6.0	3.7	3.2	12.0	3.7	2.4	8.8	0.0	0.0	0.0	0.6	0.6	0.0
Incomap	upper	in19.1b	4.5	24.1	16.0	59.8	9.6	4.7	17.6	0.0	0.0	0.0	9.6	3.8	0.0
Incomap	upper	in_Hostel		25.9	21.4	80.1	8.5	5.5	20.6	0.0	0.0	0.0	2.7	1.0	0.0
Incomap	upper	in14.1ri	6.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Incomap	upper	in14.0ru	6.0	16.8	19.2	71.8	8.4	7.1	26.4	0.0	0.0	0.0	0.4	0.2	0.0
Incomap	upper	in13.9b	6.0	21.1	3.0	11.4	10.5	7.1	26.4	0.0	0.0	0.0	2.6	1.3	0.0
Incomap	upper	in_Sable		16.5	14.2	53.0	8.3	6.5	24.4	0.0	0.0	0.0	0.9	0.4	0.0
Incomap	upper	in9.57ri		0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Incomap	upper	in9.53ru		57.7	33.2	124	16.5	12.2	45.6	0.0	0.0	0.0	1.0	0.3	0.0
Incomap	upper	in9.04b		4.0	3.2	12.0	4.0	3.0	11.4	4.0	2.8	10.4	0.4	0.4	0.4
Incomap	upper	in_Menhinick		24.8	14.4	53.8	7.5	5.5	20.7	0.5	0.4	1.4	0.5	0.2	0.1
Incomap	upper	in5.9ri	5.5	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Incomap	upper	in5.95ru	5.5	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Incomap	upper	in5.94b	5.5	3.4	3.2	12.0	10.1	9.1	34.2	3.4	2.8	10.4	0.3	0.9	0.3
Incomap	upper	in_Poole		0.6	0.6	2.2	1.9	1.7	6.3	0.6	0.5	1.9	0.1	0.2	0.1
Incomap	lower	in1.9ri	5.4	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Incomap	lower	in1.4ru	5.4	21.3	3.0	11.4	0.0	0.0	0.0	0.0	0.0	0.0	0.4	0.0	0.0
Incomap	lower	in1.8b	5.4	8.4	1.0	3.8	0.0	0.0	0.0	0.0	0.0	0.0	0.6	0.0	0.0
Incomap	lower	in_Near_mouth		16.1	2.3	8.6	0.0	0.0	0.0	0.0	0.0	0.0	0.3	0.0	0.0
Boyd		boy1	3	36.4	4.6	17.0	12.1	7.1	26.6	4.0	3.0	11.4	3.3	1.1	0.4
Boyd		boy2	3	56.6	7.1	26.5	8.1	4.7	17.7	0.0	0.0	0.0	4.4	0.6	0.0
Kelly		ke1	2.5	9.5	7.1	26.6	3.2	3.0	11.4	0.0	0.0	0.0	1.2	0.4	0.0
Kelly		ke2	2.5	60.6	35.6	133	12.1	9.1	34.2	0.0	0.0	0.0	4.9	1.0	0.0
Lex		lex1		3.9	2.4	8.9	0.0	0.0	0.0	0.0	0.0	0.0	1.1	0.0	0.0
Men		men1		11.0	7.1	26.6	11.0	9.1	34.2	0.0	0.0	0.0	1.5	1.5	0.0
Pool		po1		0.0	0.0	0.0	8.1	6.1	22.8	0.0	0.0	0.0	0.0	0.8	0.0
Pool		po2		8.6	4.7	17.7	8.6	6.1	22.8	0.0	0.0	0.0	1.1	1.1	0.0
Sable		sa1		22.8	19.0	70.9	14.2	15.2	57.0	2.8	2.6	9.6	2.0	1.2	0.2
Sable		sa2		31.1	21.3	79.8	6.9	6.1	22.8	0.0	0.0	0.0	2.7	0.6	0.0
Illecillew	upper	il17.9ri		12.1	1.5	5.7	4.0	2.4	8.9	4.0	4.3	16.0	0.4	0.1	0.1
Illecillew	upper	il18.1ru		23.6	3.5	13.2	16.8	11.9	44.3	6.7	8.5	31.9	1.2	0.8	0.3
Illecillew	upper	il_WestTwin		18.5	2.7	9.9	11.2	7.7	28.7	5.6	6.7	24.9	0.9	0.5	0.3
Illecillew	upper	il30.5ri	0.5	28.1	2.5	9.5	11.2	4.7	17.7	5.6	4.3	16.0	1.4	0.6	0.3
Illecillew	upper	il30.9ru	0.5	57.7	8.1	30.3	43.3	28.2	105	3.6	4.3	16.0	3.8	2.9	0.2
Illecillew	upper	il30.6b	0.5	21.1	3.0	11.4	0.0	0.0	0.0	0.0	0.0	0.0	4.4	0.0	0.0
Illecillew	upper	il_Tangier		48.2	6.2	23.0	29.1	17.8	66.5	4.5	4.3	16.0	3.5	1.9	0.3
Albert		al1	1.2	19.2	2.5	9.5	0.0	0.0	0.0	0.0	0.0	0.0	2.1	0.0	0.0
Albert		al2	1.2	8.2	1.0	3.8	0.0	0.0	0.0	0.0	0.0	0.0	0.6	0.0	0.0

Appendix 5a. Summary of estimated rainbow trout parr densities by age class and by site in Arrow Lakes Reservoir tributary reaches in 2004. Estimates are based on snorkeler counts adjusted to account for snorkeling efficiency. 95% confidence intervals for the density estimates (shown for fish/100m^2 only) reflect error in the estimation of snorkeling efficiency.

			Fish/100m												Fish/100m^2			
			Age 1+			Age 2+			Age 3+			Age 4+			1+	2+	3+	4+ &
Stream	Reach	Site	est.	LCI	UCI	est.	LCI	UCI	est.	LCI	UCI	est.	LCI	UCI	est.	est.	est.	>
Burton		bu1	68.6	50.1	109	20.0	-	-	0.0	-	-	0.0	-	-	7.1	2.1	0.0	0.0
Burton		bu2	105	76.6	166	24.6	-	-	0.0	-	-	0.0	-	-	9.0	2.1	0.0	0.0
Caribou	lower	car1	142	104	226	25.2	-	-	0.0	-	-	0.0	-	-	10.9	1.9	0.0	0.0
Caribou	lower	car2	216	158	343	40.1	-	-	0.0	-	-	0.0	-	-	11.2	2.1	0.0	0.0
Caribou	upper	car3	32.2	23.5	51.0	21.7	-	-	0.0	-	-	10.4	7.6	16.5	2.0	1.4	0.0	0.7
Caribou	upper	car4	52.1	38.1	82.7	25.4	-	-	0.0	-	-	9.5	6.9	15.0	4.1	2.0	0.0	0.7
Greeley		gr1	0.0	-	-	0.0	-	-	0.0	-	-	3.0	2.2	4.7	0.0	0.0	0.0	0.3
Greeley		gr2	0.0	-	-	1.7	-	-	0.0	-	-	0.0	-	-	0.0	0.2	0.0	0.0
Halfway	lower	ha1	464	339	736	116	10.0	21.7	13.7	10.0	21.7	0.0	-	-	22.5	5.6	0.7	0.0
Halfway	lower	ha11	194	141	307	51.9	2.5	5.5	3.5	2.5	5.5	0.0	-	-	10.4	2.8	0.2	0.0
Halfway	lower	ha10	212	155	337	104	63.8	139	87.4	63.8	139	6.3	4.6	9.9	10.8	5.3	4.4	0.3
Halfway	lower	ha2	239	175	379	92.9	20.0	43.3	27.3	20.0	43.3	0.0	-	-	14.1	5.5	1.6	0.0
Halfway	upper	ha3	0.0	-	-	10.9	12.0	26.0	16.4	12.0	26.0	21.9	16.0	34.7	0.0	0.7	1.0	1.3
Halfway	upper	ha4	8.7	6.4	13.9	35.5	18.0	39.0	24.6	18.0	39.0	18.8	13.7	29.7	0.5	2.0	1.4	1.1
Halfway	upper	ha5	13.4	9.8	21.2	26.0	14.0	30.3	19.1	14.0	30.3	14.1	10.3	22.3	0.7	1.3	1.0	0.7
Halfway	upper	ha6	0.0	-	-	0.0	10.0	21.7	13.7	10.0	21.7	21.9	16.0	34.7	0.0	0.0	0.8	1.3
Jordan		jr1	0.0	-	-	0.0	-	-	0.0	-	-	0.0	-	-	0.0	0.0	0.0	0.0
Jordan		jr2	0.0	-	-	0.0	-	-	0.0	-	-	0.0	-	-	0.0	0.0	0.0	0.0
Jordan		jr3	0.0	-	-	0.0	-	-	0.0	-	-	0.0	-	-	0.0	0.0	0.0	0.0
Kuskanax		ku1	85.5	62.4	136	57.5	-	-	0.0	-	-	0.0	-	-	3.3	2.2	0.0	0.0
Kuskanax		ku2	117	85.7	186	69.8	7.3	15.8	10.0	7.3	15.8	0.0	-	-	6.4	3.8	0.5	0.0
Kuskanax		ku3	29.5	21.5	46.8	49.2	11.4	24.8	15.6	11.4	24.8	46.9	34.2	74.3	1.3	2.2	0.7	2.1
Macdld.	lower	mc1	37.9	27.7	60.1	16.2	-	-	0.0	-	-	0.0	-	-	3.8	1.6	0.0	0.0
Macdld.	lower	mc2	63.9	46.7	101	56.7	-	-	0.0	-	-	2.9	2.2	4.7	9.1	8.1	0.0	0.4
Macdld.	upper	mc3	15.4	11.3	24.5	2.4	-	-	0.0	-	-	4.1	3.0	6.5	2.7	0.4	0.0	0.7
Macdld.	upper	mc4	34.7	25.4	55.1	1.8	-	-	0.0	-	-	0.0	-	-	7.1	0.4	0.0	0.0
Taite		ta1	67.0	48.9	106	92.8	8.6	18.7	11.8	8.6	18.7	2.9	2.2	4.7	10.0	13.9	1.8	0.4
Taite		ta2	35.6	26.0	56.5	32.7	2.5	5.4	3.4	2.5	5.4	0.0	-	-	5.6	5.2	0.5	0.0
Taite		ta3	64.3	46.9	102	39.8	11.9	25.8	16.3	11.9	25.8	6.5	4.8	10.3	10.7	6.6	2.7	1.1

Appendix 5b. Summary of estimated rainbow trout parr densities by age class and by site in Arrow Lakes Reservoir tributary reaches in 2005. Estimates are based on snorkeler counts adjusted to account for snorkeling efficiency. Shaded rows show mean density values for Illecillewaet River sampling locations, which consisted of three sites in three different habitat types (run/pool, riffle, side-channel/braid).

			Fish/100m												Fish/100m^2			
			Age 1+			Age 2+			Age 3+			Age 4+			1+	2+	3+	4+ &
Stream	Reach	Site	est.	LCI	UCI	est.	LCI	UCI	est.	LCI	UCI	est.	LCI	UCI	est.	est.	est.	>
Caribou	lower	car1	38.9	28.4	61.6	18.4	13.4	29.2	0.0	-	-	0.0	-	-	2.7	1.3	0.0	0.0
Caribou	lower	car2	106	77.0	167	38.4	28.0	60.8	5.4	3.9	8.5	0.0	-	-	6.5	2.3	0.3	0.0
Caribou	upper	car3	4.8	3.5	7.6	12.7	9.3	20.1	1.3	1.0	2.1	11.7	8.6	18.6	0.3	0.9	0.1	0.9
Caribou	upper	car3.5	9.6	7.0	15.2	31.0	22.7	49.2	3.1	2.3	5.0	20.3	14.8	32.2	0.7	2.1	0.2	1.4
Caribou	upper	car4	5.5	4.0	8.7	41.8	30.5	66.2	12.5	9.1	19.8	12.5	9.1	19.8	0.5	4.1	1.2	1.2
Caribou	upper	car4.5	44.2	32.3	70.1	39.4	28.8	62.5	12.5	9.1	19.8	15.6	11.4	24.8	3.1	2.8	0.9	1.1
Greeley		gr1	1.3	1.0	2.1	0.0	-	-	0.0	-	-	0.0	-	-	0.2	0.0	0.0	0.0
Greeley		gr2	6.5	4.8	10.4	3.0	2.2	4.7	1.7	1.2	2.7	0.0	-	-	0.8	0.4	0.2	0.0
Halfway	lower	ha1	182	133	289	46.2	33.7	73.2	57.7	42.2	91.5	13.2	9.6	20.9	8.5	2.1	2.7	0.6
Halfway	lower	ha2	78.1	57.0	124	48.4	35.4	76.8	6.9	5.1	11.0	0.0	-	-	4.1	2.6	0.4	0.0
Halfway	lower	ha4	343	250	544	120	87.6	190	35.1	25.6	55.6	6.3	4.6	10.0	18.5	6.5	1.9	0.3
Halfway	lower	ha12	59.8	43.6	94.8	56.7	41.4	89.9	25.8	18.8	40.9	11.8	8.6	18.7	4.7	4.5	2.0	0.9
Halfway	upper	ha6	11.5	8.4	18.2	10.9	7.9	17.2	10.9	7.9	17.2	12.4	9.1	19.7	0.8	0.7	0.7	0.8
Halfway	upper	ha7	46.4	33.9	73.6	15.2	11.1	24.1	22.8	16.6	36.1	26.0	19.0	41.3	2.4	0.8	1.2	1.3
Halfway	upper	ha16	0.0	-	-	3.0	2.2	4.7	3.0	2.2	4.7	20.4	14.9	32.3	0.0	0.3	0.3	1.9
Halfway	upper	ha17A	0.0	-	-	0.0	-	-	3.4	2.5	5.4	27.3	20.0	43.4	0.0	0.0	0.3	2.2
Halfway	upper	ha17B	0.0	-	-	4.1	3.0	6.6	4.1	3.0	6.6	4.7	3.5	7.5	0.0	0.3	0.3	0.4
Halfway	upper	ha18	1.1	0.8	1.7	1.1	0.8	1.7	2.2	1.6	3.5	18.8	13.7	29.7	0.1	0.1	0.3	2.2
Illecillew	lower	il2.0sc	0.0	-	-	0.0	-	-	0.0	-	-	0.0	-	-	0.0	0.0	0.0	0.0
Illecillew	lower	il2.1ru	0.0	-	-	0.0	-	-	0.0	-	-	0.0	-	-	0.0	0.0	0.0	0.0
Illecillew	lower	il2.4ri	4.0	2.9	6.4	12.6	9.2	20.0	0.0	-	-	0.0	-	-	0.1	0.3	0.0	0.0
Illecillew	lower	il_Revel	2.1	1.6	3.4	6.7	4.9	10.6	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.2	0.0	0.0
Illecillew	upper	il8.8ru	0.0	-	-	23.1	16.8	36.6	3.6	2.6	5.6	7.1	5.2	11.3	0.0	0.6	0.1	0.2
Illecillew	upper	il9.3sc	10.5	7.7	16.7	5.3	3.8	8.3	0.0	-	-	0.0	-	-	1.3	0.6	0.0	0.0
Illecillew	upper	il9.9ri	3.4	2.5	5.4	3.4	2.5	5.4	0.0	-	-	7.8	5.7	12.4	0.1	0.1	0.0	0.3
Illecillew	upper	il_Greeley	3.1	2.2	4.9	15.2	11.1	24.1	2.0	1.5	3.2	7.4	5.4	11.8	0.1	0.4	0.1	0.2
Illecillew	upper	il23.7ri	0.0	-	-	0.0	-	-	0.0	-	-	0.0	-	-	0.0	0.0	0.0	0.0
Illecillew	upper	il23.7sc	0.0	-	-	0.0	-	-	0.0	-	-	0.0	-	-	0.0	0.0	0.0	0.0
Illecillew	upper	il23.8ru	6.4	4.7	10.1	0.0	-	-	0.0	-	-	0.0	-	-	0.3	0.0	0.0	0.0
Illecillew	upper	il_Skunk	3.1	2.2	4.8	0.0	-	-	0.0	-	-	0.0	-	-	0.1	0.0	0.0	0.0
Illecillew	upper	il27.1ri	0.0	-	-	4.7	3.4	7.5	0.0	-	-	10.8	7.9	17.1	0.0	0.2	0.0	0.5
Illecillew	upper	il27.1ru	0.0	-	-	0.0	-	-	0.0	-	-	22.3	16.3	35.4	0.0	0.0	0.0	1.0
Illecillew	upper	il27.3sc	0.0	-	-	0.0	-	-	0.0	-	-	0.0	-	-	0.0	0.0	0.0	0.0
Illecillew	upper	il_Tunnel	0.0	0.0	0.0	2.5	1.8	3.9	0.0	0.0	0.0	16.3	11.9	25.8	0.0	0.1	0.0	0.7
Macdld.	lower	mc1	48.1	35.1	76.3	39.0	28.5	61.9	2.5	1.8	3.9	5.0	3.6	7.9	5.3	4.3	0.3	0.5
Macdld.	lower	mc2	25.8	18.8	40.9	35.6	26.0	56.4	6.4	4.7	10.2	8.6	6.3	13.6	3.7	5.1	0.9	1.2
Macdld.	upper	mc3	3.0	2.2	4.8	8.2	6.0	13.0	10.9	8.0	17.3	7.8	5.7	12.4	0.5	1.4	1.9	1.4
Macdld.	upper	mc3.5	0.0	-	-	4.2	3.1	6.6	0.0	-	-	1.6	1.2	2.5	0.0	0.6	0.0	0.2
Macdld.	upper	mc4	0.0	-	-	9.6	7.0	15.2	0.0	-	-	0.0	-	-	0.0	1.9	0.0	0.0
Macdld.	upper	mc4.5	0.0	-	-	12.3	9.0	19.5	0.0	-	-	0.0	-	-	0.0	2.4	0.0	0.0
Tangier	lower	tan1	0.0	-	-	0.0	-	-	0.0	-	-	0.0	-	-	0.0	0.0	0.0	0.0
Tangier	upper	tan2	0.0	-	-	0.0	-	-	0.0	-	-	0.0	-	-	0.0	0.0	0.0	0.0
Tangier	upper	tan3	0.0	-	-	0.0	-	-	0.0	-	-	0.0	-	-	0.0	0.0	0.0	0.0

Appendix 5c. Summary of estimated rainbow trout parr densities by age class and by site in Arrow Lakes Reservoir tributary reaches in 2006. Estimates are based on snorkeler counts adjusted to account for snorkeling efficiency. Shaded rows show mean density values for Illecillewaet and Incomappleaux sampling locations, which consisted of 2-5 sites representing different habitat types (run/pool, riffle, side-channel/braid).

			Fish/100m												Fish/100m^2			
			Age 1+			Age 2+			Age 3+			Age 4+			1+	2+	3+	4+ &
Stream	Reach	Site	est.	LCI	UCI	est.	LCI	UCI	est.	LCI	UCI	est.	LCI	UCI	est.	est.	est.	>
Incom	upper	in27.3ri	0.0	-	-	0.0	-	-	0.0	-	-	0.0	-	-	0.0	0.0	0.0	0.0
Incom	upper	in27.4ru	0.0	-	-	0.0	-	-	0.0	-	-	0.0	-	-	0.0	0.0	0.0	0.0
Incom	upper	in_BattleBrook	0.0	-	-	0.0	-	-	0.0	-	-	0.0	-	-	0.0	0.0	0.0	0.0
Incom	upper	in23.9ri	0.0	-	-	0.0	-	-	0.0	-	-	0.0	-	-	0.0	0.0	0.0	0.0
Incom	upper	in22.3ru	0.0	-	-	0.0	-	-	0.0	-	-	0.0	-	-	0.0	0.0	0.0	0.0
Incom	upper	in24.0ru	0.0	-	-	0.0	-	-	0.0	-	-	0.0	-	-	0.0	0.0	0.0	0.0
Incom	upper	in23.8b2	0.0	-	-	0.0	-	-	0.0	-	-	0.0	-	-	0.0	0.0	0.0	0.0
Incom	upper	in23.9b	0.0	-	-	0.0	-	-	0.0	-	-	0.0	-	-	0.0	0.0	0.0	0.0
Incom	upper	in_McDougal	0.0	-	-	0.0	-	-	0.0	-	-	0.0	-	-	0.0	0.0	0.0	0.0
Incom	upper	in17.4ri	0.0	-	-	0.0	-	-	0.0	-	-	0.0	-	-	0.0	0.0	0.0	0.0
Incom	upper	in17.5ru	0.0	-	-	0.0	-	-	0.0	-	-	0.0	-	-	0.0	0.0	0.0	0.0
Incom	upper	in19.1ru	0.0	-	-	0.0	-	-	0.0	-	-	0.0	-	-	0.0	0.0	0.0	0.0
Incom	upper	in17.4b	0.0	-	-	0.0	-	-	0.0	-	-	0.0	-	-	0.0	0.0	0.0	0.0
Incom	upper	in19.1b	0.0	-	-	0.0	-	-	0.0	-	-	0.0	-	-	0.0	0.0	0.0	0.0
Incom	upper	in_Hostel	0.0	-	-	0.0	-	-	0.0	-	-	0.0	-	-	0.0	0.0	0.0	0.0
Incom	upper	in14.1ri	0.0	-	-	0.0	-	-	0.0	-	-	0.0	-	-	0.0	0.0	0.0	0.0
Incom	upper	in14.0ru	0.0	-	-	0.0	-	-	0.0	-	-	0.0	-	-	0.0	0.0	0.0	0.0
Incom	upper	in13.9b	0.0	-	-	0.0	-	-	0.0	-	-	0.0	-	-	0.0	0.0	0.0	0.0
Incom	upper	in_Sable	0.0	-	-	0.0	-	-	0.0	-	-	0.0	-	-	0.0	0.0	0.0	0.0
Incom	upper	in9.57ri	0.0	-	-	0.0	-	-	0.0	-	-	0.0	-	-	0.0	0.0	0.0	0.0
Incom	upper	in9.53ru	0.0	-	-	0.0	-	-	0.0	-	-	0.0	-	-	0.0	0.0	0.0	0.0
Incom	upper	in9.04b	0.0	-	-	0.0	-	-	0.0	-	-	0.0	-	-	0.0	0.0	0.0	0.0
Incom	upper	in_Menhinick	0.0	-	-	0.0	-	-	0.0	-	-	0.0	-	-	0.0	0.0	0.0	0.0
Incom	upper	in5.9ri	0.0	-	-	0.0	-	-	0.0	-	-	0.0	-	-	0.0	0.0	0.0	0.0
Incom	upper	in5.95ru	0.0	-	-	0.0	-	-	0.0	-	-	0.0	-	-	0.0	0.0	0.0	0.0
Incom	upper	in5.94b	0.0	-	-	0.0	-	-	0.0	-	-	0.0	-	-	0.0	0.0	0.0	0.0
Incom	upper	in_Poole	0.0	-	-	0.0	-	-	0.0	-	-	0.0	-	-	0.0	0.0	0.0	0.0
Incom	lower	in1.9ri	0.0	-	-	6.3	-	-	0.0	-	-	0.0	-	-	0.0	0.2	0.0	0.0
Incom	lower	in1.4ru	0.0	-	-	0.0	-	-	0.0	-	-	0.0	-	-	0.0	0.0	0.0	0.0
Incom	lower	in1.8b	14.2	10.0	21.7	15.0	10.6	22.9	6.5	4.0	8.7	9.8	6.8	14.9	1.0	1.1	0.5	0.7
Incom	lower	in_Bottom	0.6	0.4	0.9	2.2	1.6	3.5	0.3	0.2	0.4	0.4	0.3	0.6	0.0	0.1	0.0	0.0
Boyd		boy1	0.0	-	-	0.0	-	-	0.0	-	-	0.0	-	-	0.0	0.0	0.0	0.0
Boyd		boy2	0.0	-	-	0.0	-	-	0.0	-	-	0.0	-	-	0.0	0.0	0.0	0.0
Kelly		ke1	0.0	-	-	0.0	-	-	0.0	-	-	0.0	-	-	0.0	0.0	0.0	0.0
Kelly		ke2	0.0	-	-	0.0	-	-	0.0	-	-	0.0	-	-	0.0	0.0	0.0	0.0
Lexing		lex1	0.0	-	-	0.0	-	-	0.0	-	-	0.0	-	-	0.0	0.0	0.0	0.0
Menhin		men1	0.0	-	-	0.0	-	-	0.0	-	-	0.0	-	-	0.0	0.0	0.0	0.0
Pool		po1	0.0	-	-	0.0	-	-	0.0	-	-	0.0	-	-	0.0	0.0	0.0	0.0
Pool		po2	0.0	-	-	0.0	-	-	0.0	-	-	0.0	-	-	0.0	0.0	0.0	0.0
Sable		sa1	0.0	-	-	0.0	-	-	0.0	-	-	0.0	-	-	0.0	0.0	0.0	0.0
Sable		sa2	0.0	-	-	0.0	-	-	0.0	-	-	0.0	-	-	0.0	0.0	0.0	0.0
Illecillew	upper	il30.5ri	0.0	-	-	0.0	-	-	0.0	-	-	0.0	-	-	0.0	0.0	0.0	0.0
Illecillew	upper	il30.9ru	0.0	-	-	0.0	-	-	0.0	-	-	0.0	-	-	0.0	0.0	0.0	0.0
Illecillew	upper	il30.6b	0.0	-	-	0.0	-	-	0.0	-	-	0.0	-	-	0.0	0.0	0.0	0.0
Illecillew	upper	il_Tangier	0.0	-	-	0.0	-	-	0.0	-	-	0.0	-	-	0.0	0.0	0.0	0.0
Illecillew	upper	il17.9ri	0.0	-	-	2.7	-	-	3.1	-	-	0.0	-	-	0.0	0.1	0.1	0.0
Illecillew	upper	il18.1ru	0.0	-	-	0.0	-	-	2.6	-	-	2.6	-	-	0.0	0.0	0.1	0.1
Illecillew	upper	il_WestTwin	0.0	-	-	1.2	-	-	2.8	-	-	1.5	-	-	0.0	0.0	0.1	0.1
Albert		al1	0.0	-	-	0.0	-	-	0.0	-	-	0.0	-	-	0.0	0.0	0.0	0.0
Albert		al2	0.0	-	-	0.0	-	-	0.0	-	-	0.0	-	-	0.0	0.0	0.0	0.0

Appendix 6. Summary of unadjusted snorkeler counts, by site, for bull trout and rainbow trout fry and adults, mountain whitefish adults and juveniles, and eastern brook trout in Arrow Lakes Reservoir tributary reaches during 2004-2006. For adult kokanee, longnose dace, sculpin (*Cottid spp.*), and redisided shiner, P indicates that at least one fish was observed by snorkelers at the site.

				Bull trout			Rainbow trout		Whitefish		Eastern				
Year	Stream	Reach	No. of sites	Adfluvial adult (400-800 mm)	4+ or older resident (210-350 mm)	Age-0+ fry	4+ or older resident (190-300 mm)	Age-0+ fry	Adult or sub-adult (200-300 mm)	Juvenile (60-200 mm)	brook trout (125-220 mm)	Adult kokanee	Long-nose dace	Cottid spp.	Red-sided shiner
2004	Burton		2	0	0	0	0	6	1	4	0	P	P		
2004	Caribou	lower	2	1	0	2	0	37	12	89	0	P	P		
2004	Caribou	upper	2	3	0	8	5	1	0	0	1				
2004	Greeley		2	3	1	10	2	0	0	0	0				
2004	Halfway	lower	4	0	0	1	2	67	6	8	0	P			
2004	Halfway	upper	4	4	0	3	29	4	0	0	0	P		P	
2004	Jordan	lower	3	1	0	0	0	0	1	22	1	P		P	
2004	Kuskanax	lower	2	0	0	0	0	20	8	48	0	P	P	P	
2004	Macdld.	lower	2	2	0	1	1	15	0	8	0	P			
2004	Macdld.	upper	2	5	5	10	3	0	0	0	0				
2004	Taite		3	0	0	1	3	9	0	1	0	P			
2005	Caribou	lower	2	0	0	0	0	57	13	83	0	P	P		
2005	Caribou	upper	4	0	0	7	31	8	0	0	2				
2005	Greeley		2	5	2	15	0	0	0	0	0				
2005	Halfway	lower	4	0	0	0	9	81	4	27	0	P		P	
2005	Halfway	upper	6	0	11	1	52	9	0	0	0			P	
2005	Illecillew	lower	3	0	1	0	0	0	2	21	0	P		P	P
2005	Illecillew	upper	10	0	1	39	12	2	0	267	0			P	
2005	Macdld.	lower	2	0	0	1	6	23	2	16	0	P			
2005	Macdld.	upper	4	27	0	11	6	2	0	0	0				
2005	Tangier	lower	1	0	1	3	0	0	0	0	0				
2005	Tangier	upper	2	0	6	1	0	0	0	0	0				
2006	Incom	lower	3	0	1	2	0	30	0	1	0			P	
2006	Incom	upper	21	0	3	21	0	0	4	125	0			P	P
2006	Incom Tribs		10	0	2	51	0	0	0	3	0			P	P
2006	Illecillew	upper	5	0	2	17	0	0	0	18	0			P	
2006	Albert		2	0	0	3	0	0	0	0	0				

Appendix 7. UTM coordiantes and upstream distances from the Arrow Lakes Reservoir for barriers (upstream limit to migration for all fish) and major obstructions in the study tributaries. Obstructions likely limit upstream migration for all fish except adfluvial adult bull trout, and were used to delinate lower and upper reaches in streams where they occurred.

		Upstream barrier			Major obstruction		
Principal watershed	Tributary	UTM Easting	UTM Northing	Distance from stream mouth (km)	UTM Easting	UTM Northing	Distance from stream mouth (km)
Burton/Caribou	Burton	n/a	n/a	n/a	-	-	-
Burton/Caribou	Caribou	441088	5539384	5.9	440162	5537537	4.1
Halfway	Halfway	n/a	n/a	n/a	443832	5595009	9.9
Kuskanax	Kuskanax	447694	5570383	8.5	-	-	-
Macdonald	Macdonald	452267	5550181	11.5	-	-	-
Taite	Taite	410538	5518127	2.2	-	-	-
Jordan	Jordan	412035	5653926	2.9	-	-	-
Illecillewaet	Illecillewaet	441574	5666259	39.0	418385	5648980	4.2
Illecillewaet	Albert	439242	5663531	2.7	-	-	-
Illecillewaet	Greely	425802	5651225	1.8	-	-	-
Illecillewaet	Tangier	440734	5667049	1.3	-	-	-
Illecillewaet	Twin	433038	5654622	0.4	-	-	-
Illecillewaet	Woolsey	437202	5663381	0.9	-	-	-
Incomappleaux	Incomappleaux	458817	5650928	39.6	452926	5625708	5.4
Incomappleaux	Battle Brook	459455	5650060	0.1			
Incomappleaux	Boyd	459883	5638417	1.8	-	-	-
Incomappleaux	Kelly	459987	5640999	1.6	-	-	-
Incomappleaux	Lexington	456458	5632475	0.6	-	-	-
Incomappleaux	McDougal	457620	5644159	0.2	-	-	-
Incomappleaux	Menhenick	454899	5629028	0.5	-	-	-
Incomappleaux	Pool	454993	5626779	1.2	-	-	-
Incomappleaux	Sable	454932	5633669	1.9	-	-	-

Appendix 8. Summary of low level element concentrations in water samples collected in the ALR and selected tributaries in September 2006.

	Units	ALR-NARR OWS 1	ALR-NARR OWS 2	ALR-REVEL 1	ALR-REVEL 2	HALF 1	HALF 2	MAC 1	MAC 2	CARIB 1	CARIB 2	INCOM 1	INCOM 2	ILLEC 1	ILLEC 2
Total Mercury (Hg)	ug/L	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
Total Aluminum (Al)	mg/L	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
Total Barium (Ba)	mg/L	0.01	0.01	0.02	0.02	0.04	0.04	0.01	0.01	0.02	0.01	0.02	0.02	0.01	0.03
Total Beryllium (Be)	mg/L	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
Total Bismuth (Bi)	mg/L	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
Total Boron (B)	mg/L	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
Total Calcium (Ca)	mg/L	14	14.2	16.6	17.2	5.2	5.4	36.6	37.1	16.8	16.9	24	24.2	7.5	15.8
Total Chromium (Cr)	mg/L	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
Total Copper (Cu)	mg/L	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
Total Iron (Fe)	mg/L	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	0.06	ND	ND
Total Magnesium (Mg)	mg/L	3.1	3.3	4.7	4.8	0.5	0.5	6	6.2	1.7	1.6	2.5	2.5	1.5	2.7
Total Manganese (Mn)	mg/L	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	0.02	0.01	ND	ND
Total Molybdenum (Mo)	mg/L	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
Total Nickel (Ni)	mg/L	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
Total Phosphorus (P)	mg/L	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
Total Potassium (K)	mg/L	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
Total Silicon (Si)	mg/L	1.1	1.2	1.4	1.4	3.1	3.1	4	4	4	3.8	2	1.9	4	3.3
Total Sodium (Na)	mg/L	ND	ND	ND	ND	ND	0.5	1.3	1.6	1.1	1.1	0.8	0.8	1.8	2.3
Total Strontium (Sr)	mg/L	0.09	0.09	0.11	0.11	0.39	0.39	0.24	0.24	0.08	0.08	0.15	0.15	0.05	0.07
Total Sulphur (S)	mg/L	3	3	4	4	3	3	6	6	3	3	3	3	2	3
Total Tin (Sn)	mg/L	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
Total Titanium (Ti)	mg/L	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
Total Vanadium (V)	mg/L	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
Total Zinc (Zn)	mg/L	ND	0.06	0.13	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
Total Zirconium (Zr)	mg/L	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
Total Antimony (Sb)	ug/L	ND	ND	ND	ND	ND	1	2	ND	ND	ND	ND	ND	ND	ND
Total Arsenic (As)	ug/L	ND	1	1	ND	1	1	2	2	2	2	ND	ND	ND	ND
Total Cadmium (Cd)	ug/L	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
Total Cobalt (Co)	ug/L	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
Total Lead (Pb)	ug/L	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
Total Selenium (Se)	ug/L	ND	2	1	ND	2	ND	2	2	2	2	ND	ND	ND	ND
Total Silver (Ag)	ug/L	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
Total Thallium (Tl)	ug/L	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
Total Uranium (U)	ug/L	0.3	0.3	0.3	0.3	0.1	0.1	0.3	0.3	0.4	0.4	1.2	1.2	0.5	0.8

Appendix 9. Photographic plates 1-4.

Plate 1. Major obstruction on Halfway River 9.9 km from the stream mouth.

Plate 2. Example of a bull trout parr with a custom-made chenille tag.

Plate 3. Major obstruction on Incomappleaux River 5 km from the stream mouth. A large adfluvial bull trout can be seen attempting the falls in the middle of the frame.

Plate 4. Typical spawning habitat in Section 1 of upper Halfway River (see map in Figure 1c).