

**THE DISTRIBUTION AND ABUNDANCE OF JUVENILE
CHINOOK SALMON ABUNDANCE IN THE LOWER THOMPSON
RIVER BASIN IN RELATION TO SPAWNER ABUNDANCE AND
HABITAT CHARACTERISTICS**

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ABSTRACT

As part of a long-term goal to rebuild Fraser River Chinook salmon *Oncorhynchus tshawytscha* stocks, the Pacific Salmon Commission has adopted the concept of using maximum sustained yield (MSY) or other biologically-based escapement goals for stock groups consisting of individual streams, whereas DFO's Wild Salmon Policy relies on benchmarks to assess the biological status for conservation units. Currently, adult-to-adult stock-recruitment data are of limited utility for estimating stock-recruitment relationships for interior Fraser River Chinook populations because of insufficient coded wire tag application and fishery recovery data required to estimate exploitation rates for each stock. Assuming density independent fry-to-adult survival, estimates of total fry standing stock for individual streams are a potentially useful alternative index of recruitment. During 2001-2006, we evaluated methodologies for estimating Chinook fall fry standing stocks in tributaries of the lower Thompson basin (below Kamloops Lake) and in the lower Thompson River itself. We used a novel mark-recapture methodology to evaluate the accuracy of night snorkel counts as a population estimation methodology for deeper habitats (runs and pools in smaller tributaries, and all habitat types in most larger stream reaches), and evaluated single-pass electrofishing for shallow riffle habitats in smaller tributaries by comparing first-pass catches to maximum likelihood estimates derived from multiple-pass electrofishing. Estimates of mean snorkeling efficiency (proportion of Peterson mark-recapture estimates seen by snorkelers; 53%) and first-pass electrofishing efficiency (78%) were reasonably precise (95% confidence intervals: 48-59% and 63-93%, respectively) and without obvious outliers, suggesting that both methods provided reliable estimates of fry standing stock when adjusted for sampling efficiency.

In each year of the study we applied the rapid assessment methods to basin-wide, stratified systematic surveys of Chinook fry abundance. Chinook fry density showed substantial variability, both spatially across the study area and among years, but there was a consistent pattern of increasing fry densities from upstream to downstream reaches, both within individual streams, and within the study area as a whole. Fry were also more

abundant in relatively deep, lower velocity habitats, and in lower gradient stream sections. Marginal habitats that supported lower fry densities experienced greater interannual variation in fry abundance compared to core (high quality) rearing habitats. Our data also suggested density dependent growth for Chinook fry in Thompson River tributaries at recent fry abundances.

We examined adult-to-fry stock-recruitment relationships for individual tributaries using our 2001-2006 data, and data from prior studies. Although the contrast of brood year spawner densities and the number of stock-recruitment data points were insufficient to reliably estimate stock-recruitment parameters (S_{msy} and S_{rep}), large per unit area differences among tributaries in carrying capacity and in fry recruit-per-spawner relationships were evident. For some tributaries, stock-recruitment analysis indicated density-dependent mortality, suggesting that recent high escapements may have approached or even exceeded S_{msy} . In other tributaries, a linear relationship between spawner and fry abundance indicated little density dependent competition and suggested recent escapements were considerably below S_{msy} . Evaluation of escapement goals and methods would benefit from higher escapements to increase the contrast in the spawner-fry production relationship and more extensive juvenile Chinook surveys to assess underestimation biases in our standing stock estimates (i.e. sampling from Nicola Lake to Coldwater River confluence or Fraser River mainstem downstream of Lytton).

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1.0 INTRODUCTION

1.1 Developing escapement goals for stream-type interior Fraser Chinook

The Fraser River system is the largest producer of Chinook salmon (*Oncorhynchus tshawytscha*) in Canada (Northcote and Larkin 1989; Bradford and Taylor 1997), and Chinook adults homing to natal streams in the interior Fraser River watershed face interception in commercial, aboriginal and recreational fisheries. Chinook salmon escapements to Fraser River declined from the 1950's to the 1980's as a result of overfishing and other factors (CTC 1998), which led DFO to initiate a stock rebuilding and monitoring program beginning in the 1980's. Interim escapement goals for each stock or stock aggregate were set as double the average escapement from 1979-1982, years when stock abundances were depressed due to high exploitation rates (goals for some stocks were later revised to double the 1984 escapement; CTC 1998). These goals were meant to be interim and ultimately replaced by escapement goals derived from some measure of productive capacity.

The concept of using maximum sustained yield (MSY) or other biologically-based reference points (see Richards and Maguire 1998) to determine target escapements for anadromous salmonid populations has been adopted by both DFO and the Pacific Salmon Commission (PSC 2000). Currently, adult-to-adult stock-recruitment data are of limited utility for estimating stock-recruitment relationships of interior Fraser River Chinook populations because coded wire tag application and recovery data, used to estimate exploitation rates for each stock, has been insufficient (Parken et al. 2002). Ideally, smolt production would be assessed relative to adult escapement levels to determine the stock-recruitment relationship, by collecting smolt numbers in rearing habitats over a wide range of escapements. This would reduce the imprecision caused by variability in harvest and marine survival. However, interior Fraser streams experience high, snowmelt-driven streamflows during the smolt migration, which makes smolt trapping impossible or prohibitively expensive (Parken et al. 2002). Further, studies of juvenile Chinook salmon behaviour indicate a nearly continuous redistribution of fish within and emigration from

the natal stream, which makes direct measurement of smolt production logistically dubious for essentially all Fraser River Chinook salmon populations.

In the absence of juvenile recruitment data, habitat-based models have been used to predict freshwater carrying capacity and to develop escapement targets. Parken et al. (2002) reviewed patterns of juvenile Chinook behaviour and habitat use in interior Fraser systems, and suggested that the major factor limiting freshwater production was likely spawning habitat. Interior Fraser Chinook stocks disperse from spawning areas to downstream rearing habitat throughout their natal tributaries and, in some cases, to rearing habitats in nonnatal tributaries (Murray and Rosenau 1989) or in the Fraser River mainstem (Bradford and Taylor 1997). Because of this downstream dispersal and the apparent large amount of downstream rearing habitat available, Parken et al. (2002) argued that spawning habitat was a more likely bottleneck than juvenile rearing habitat. Spawning habitat limitation implies density dependence between adult spawner abundance and viable eggs remaining in the gravel following the spawning period. Based on this assumption, they developed two separate models to predict spawner capacity. The first relied on empirical data as input, with the observed maximum spawner densities in individual reaches being used as surrogates for carrying capacity. In the second approach, carrying capacity was predicted from relationships between observed maximum spawner density and stream gradient. In most cases, preliminary escapement goals developed by Parken et al. (2002) were several-fold higher than those originally proposed (Healey 1982). The assumption that spawning habitat is limited, but subsequent egg-to-fry and fry-to-smolt survival is density independent, is untested, making any such predictions of carrying capacity uncertain. For stream-rearing salmonids, density dependent survival in early life stages is typical (Kennedy and Crozier 1993; Ward and Slaney 1993; Hartman et al. 1996; Elliott 1987), and is to be expected in highly suitable, core habitats of a species range (Taylor et al. 1978; Elliott 1987). Examples of density dependent mortality agents include redd superimposition (Bonnell 1991), spawning failure such as pre-spawn mortality and egg retention (Quinn et al. 2007) competition for limited food and cover (Chapman 1966), and predation (Wood 1987).

More recently, Parken et al. (2006) developed models to predict freshwater habitat capacity from watershed area and life history type, based on stock-recruitment data from 25 Chinook salmon stocks distributed along the length of the Pacific coast between central Alaska and northern Oregon. These models assume that Chinook salmon populations may be limited by either spawning or rearing habitat, or both. At a coarse scale, freshwater habitat increases with river network size, and river network size is strongly associated with the watershed area that captures precipitation and contributes water to the channel network that drains it (Parken et al. 2006). Although, this approach represents a significant advance in estimating escapement goals for data limited stocks, no adult-to-adult stock-recruitment data were available for interior Fraser streams for use in model development. As well, during model validation the potential for substantial inaccuracy was indicated, particularly for streams with unusually small spawning areas, or small watershed areas in the case of the stream-type life history stocks (Parken et al. 2006).

The development of simple, habitat-based models to estimate stock-recruitment parameters is a useful step towards establishing escapement goals for interior Fraser Chinook stocks, but reliable empirical stock-recruitment data are needed to evaluate these models. When adult-to adult stock-recruitment data is substituted for adult-to-smolt data, variation in marine survival, life history complexity, and uncertainty about exploitation rates in commercial fisheries can mask stock-recruitment relationships or introduce substantial error. As well, for some stocks and management situations, more precise information is needed about the stock-recruitment relationship than can be obtained by habitat-based models.

While direct measures of Chinook salmon smolt production from interior Fraser streams may not be feasible, alternative indices of freshwater production may be useful in that stock-recruitment relationships could still be assessed in the absence of the confounding effect of variability in marine survival. For interior Fraser streams, the most feasible surrogate for Chinook smolt abundance is likely fall fry standing stock. Adult-

to-fall fry stock-recruitment data are potentially useful for evaluating habitat-based model predictions of stock recruitment parameters for interior Fraser Chinook stocks developed by Parken et al. (2006), and would represent a refinement of the current approach of using adult-to-adult data. Habitat-derived estimates of MSY should correspond to minimum escapement levels that still yield maximum fry standing stocks, assuming no bottlenecks to production at later life stages. The interim escapement goals of double the average escapement for 1979-1982 have been met at times for stream-type Chinook stocks in the lower Thompson basin, but how this relates to juvenile production has not been investigated. Given the variable early life history of Chinook (Bradford and Taylor 1997), fry data may also be useful in understanding the relative importance of habitats in natal versus downstream reaches to freshwater production.

The objectives of this study were to examine the feasibility of obtaining reliable estimates of Chinook fry abundance for streams in the lower Thompson River basin (downstream of Kamloops Lake, Figure 1), to investigate spatial and temporal patterns in fry production, and to assess the potential for developing stock-recruitment relationships for these streams using paired fry and spawner abundance data. Ultimately, adult-to-fry stock-recruitment relationships can be useful for evaluating existing models of freshwater carrying capacity for these streams where spawner abundance data have sufficient contrast.

1.2 Estimating juvenile Chinook abundance in large river basins

In all but the smallest streams, the reliability of population estimation methods for Chinook fry or other juvenile salmonids has received little attention (Gardiner 1984; Mitro and Zale 2002). Electrofishing, the standard method for assessing juvenile abundance in streams, has two major disadvantages with respect to basin-wide surveys and larger rivers. First, the method is very time consuming and resources are often insufficient to allow for the large number of sample sites necessary to address the typically high spatial heterogeneity in fish abundance. Stream fish populations are often highly clumped in their distribution as a result of spawning distribution (Beard and Carline 1991), geomorphic influences such as elevation, channel slope, and stream size

(Kruse et al. 1997), habitat variability (Newman and Waters 1984), intercohort competition (Bohlin 1978), channel alterations (Elser 1968), or other factors. Spatial heterogeneity in juvenile fish abundance is likely to be more pronounced at the scale of entire stream basins where variation in these factors is greater compared to smaller individual streams.

Several studies have demonstrated that uncertainty in estimates of total juvenile standing stock in small, individual streams can be reduced by applying ‘rapid assessment’ methods that sacrifice accuracy at individual sites to allow a larger number of sites to be sampled for a given amount of sampling effort (Hankin and Reeves 1988; Jones and Stockwell 1995; Wyatt 2002; Bateman et al. 2005). Inherent to this approach is the calibration of the rapid assessment method using a more accurate method. Hankin and Reeves (1988) used daytime snorkeling as a rapid assessment method and calibrated snorkeling counts by conducting multiple-pass depletion electrofishing at portion of the sampling sites. Jones and Stockwell (1995) and Bateman et al. (2005) took a similar approach, but substituted single-pass electrofishing catches for snorkeling counts. Peterson et al. (2004) improved upon the use of depletion electrofishing as a calibration method by incorporating mark-recapture methods. Abundance estimates derived from rapid assessments methods may be strongly biased if the calibration methodology itself is not accurate (Thompson 2003).

The second disadvantage of electrofishing is that it is frequently impossible to achieve good results for juvenile salmonids in larger streams. Shore-base electrofishing has been used at unenclosed sites in medium-sized streams (Mitro and Zale 2002), but mainstem habitats often cannot be enclosed with stop nets without substantial site disturbance, if they can be enclosed at all, and excessive depth and current velocity often severely reduces the effectiveness of electrofishing. Boat-based electrofishing is sometimes used to sample juvenile fish in larger streams, but the method is limited to habitats that can be navigated by boat and where netting of fish is possible. In contrast to electrofishing, snorkeling can be applied to a broader size range of streams (Hall-Griswold and Petrovsky 1996; Satterthwaite 2002), but similar to electrofishing, little work has been

done to quantify the relationship between snorkeling counts and juvenile fish abundance in larger streams (Gardiner 1984).

Obtaining reliable estimates of total juvenile standing stocks for a watershed or stream network requires a sampling method that is effective in all stream reaches and habitats that provide rearing areas for juveniles. The method must also be relatively fast to allow for adequate replication to address high spatial heterogeneity in fish abundance across a diverse study area. An accurate method of estimating abundance is also needed to calibrate count or catch data provided by the rapid assessment method. As part of a study to address steelhead (*O. mykiss*) parr carrying capacity in the lower Thompson River (Hagen et al. 2007 in prep.), during 2001-2006, we also evaluated the use of night snorkeling to estimate fall standing stocks of Chinook fry both for individual streams and the basin as a whole. Owing to low, stable flows and good to excellent water clarity during the fall in most years, most of the usable Chinook fry habitat in the lower Thompson River is readily sampled using snorkeling surveys.

Our study area included 460 km of habitat comprising a diversity of streams ranging from third-order headwater tributaries with wetted widths less than 10 m, to a high-order mainstem river with wetted widths exceeding 100 m. We tested a snorkeling calibration method for open sampling sites that was based on mark-recapture methodology and a novel capture technique that attempted to minimize the potential effects of capture and handling on the detection rate of marked parr during subsequent snorkeling surveys (snorkeling efficiency). We examined the reliability of the calibration method itself by comparing snorkeling efficiency estimates to those obtained using a second, independent method based on closed-section depletion electrofishing, in streams small enough for this second method to be feasible. The study also examined movement of marked parr from open sampling sites and the effect of sampling conditions, and physical and biological factors on snorkeling efficiency. Riffle habitats in smaller tributaries of the lower Thompson River (6 of 27 sampling strata in our study area) are too shallow for snorkeling to be effective. For these strata, single-pass electrofishing was substituted for

snorkeling, and three-pass depletion electrofishing was employed at a portion of the electrofishing sites to calibrate the single-pass catches.

2.0 METHODS

2.1 Study area

The geographic extent of our study was the major anadromous portion of the lower Thompson River watershed including 125 km of the Thompson mainstem from Kamloops Lake to the Fraser River confluence (Figure 1) and 337 km of habitat in the primary (Nicola, Bonaparte, Deadman rivers) and larger secondary tributaries (Coldwater River and Spius Creek). Tributary streams had watershed areas ranging from 800 to 7300 km² (Table 1). Juvenile Chinook have also been recorded in the upper reaches of the Nicola River (\approx 20 km of stream length, Sebastian 1982), in the lower reaches of other secondary tributaries (Nuaitch Creek: lower 7 km, Tredger 1980a; Guichon Creek: lower 5 km, Sebastian 1982; Maka Creek, lower 16 km, Sebastian 1982; Criss Creek; lower 4 km, Decker and Hagen 2004). However, we excluded these latter reaches because of either poor access or their modest contribution to overall juvenile Chinook abundance in the lower Thompson River basin.¹

Extensive scale-age analysis of return spawners indicate that Chinook stocks from most streams in the South Thompson River and those from the lower Thompson mainstem are ocean-type, rearing in freshwater for 60-150 days prior to ocean migration, whereas mainstem and tributary populations from the North Thompson and tributary populations from the lower Thompson are exclusively stream-type, spending one year (rarely two years) in freshwater prior to ocean migration (Candy et al. 2002). Analyses of spawning areas, life histories, spawner run-timing and microsatellite DNA have suggested that populations in the lower Thompson River tributaries are distinct from others in the Fraser River watershed (Candy et al. 2002). These latter populations include early spring-run populations (peak spawning in August) in the upper 51 km of the

¹ Estimated Chinook standing stock for these reaches in 1981 was about 15,000 fry, < 2% of estimated standing stocks for the lower Thompson River system during 2001-2006; Table 5).

Coldwater River (reach CW2; Figure 1) and the upper 32 km of Spius Creek (SP2) and major tributary, Maka Creek, and spring-run (peak spawning in September) populations in the Thompson River mainstem, the Deadman/Criss Creek system, the Bonaparte River, and the Nicola River (Bailey et al. 2001; Candy et al. 2002). The Nicola population also spawns in the lower 28 km of the Coldwater River and the lower 7 km of Spius Creek, (CW1 and SP1, respectively; Figure 1; Parken et al. 2003). We sampled Chinook fry abundance in the lower Thompson River mainstem as part of this study because fry present there during our annual October surveys are the progeny of spawning stocks from lower Thompson tributaries, and represent a portion of total recruitment (fry from North Thompson stream-type populations also rear in the lower Thompson River mainstem; Sections 2.7 and 2.8.1 describe how these stocks were partitioned).

The lower Thompson River watershed lies within the interior dry belt, and overlaps four biogeoclimatic zones (Krajina 1959): the semi-arid Ponderosa Pine and Bunch Grass zones in the lower elevations areas near the Thompson River mainstem (Nicola, lower Bonaparte, and Deadman rivers), and the Interior Douglas Fir and Montane Spruce zones in the wetter, higher elevation tributaries (Coldwater, upper Bonaparte and Deadman rivers, and Spius and Criss Creeks). Summers are typically hot and dry, and winters cold and dry, although the upper reaches of some tributaries receive substantial snowfalls in winter. In the tributaries the hydrograph is snowmelt driven, with peak flows during the May/June freshet followed by low summer and winter flows (Table 1), while in the Thompson River, discharge is maintained at relatively high levels until early fall due to an extensive upper watershed area that includes large lakes and numerous glacier-fed tributaries.

Other salmonids inhabiting the study area include coho (*O. kisutch*), sockeye (*O. nerka*), and pink salmon (*O. gobuscha*), as well as steelhead trout (*O. mykiss*), bull trout (*Salvelinus confluentus*), brook trout (*S. fontinalis*), and mountain whitefish (*Prosopium williamsoni*). Largescale suckers (*Catostomus macrocheilus*), longnose dace (*Rhinichthys cataractae*), prickly sculpin (*Cottus asper*), northern pikeminnow (*Ptchocheilus oregonensis*), and red sided shiner (*Richardsonius balteatus*) are also common (McPhail and Carveth 1994).

2.2 Stratification of sampling effort and site selection

We stratified juvenile Chinook population sampling by reach and habitat type. We delineated three mainstem reaches in the Thompson River and nine tributary reaches (Table 1; Figure 1). Our decisions about reach breaks in the Thompson River mainstem were based on expected differences in habitat quality for juvenile Chinook and proximity to major spawning tributaries. Reach T1 includes the 39 km section of the mainstem from Kamloops Lake to the mouth of the Bonaparte River (Table 1; Figure 1), reach T2 includes the 57 km section between the Bonaparte River and the Nicola River, and reach T3 represents the 28 km canyon section which extends from the mouth of the Nicola River downstream to the confluence of the Thompson and Fraser rivers. We divided the Coldwater River and Spius Creek into two reaches that corresponded to the spawning distributions of early spring-run populations in the upper reaches of the two streams (upper reaches, CW2 and SP2; Figure 1) and the spring-run Nicola River population that spawns in the lower reaches of these streams (CW1 and SP1). The Nicola and Bonaparte Rivers were divided into two reaches to improve sampling efficiency, since juvenile densities were reported to be higher in the lower reach of each stream compared to the upper reach (Sebastian 1982), which our sampling data confirmed. We located the uppermost reach boundary in each tributary based on the Chinook distribution described in the Fisheries Inventory Summary System (FISS, DFO 2001). The exception to this was the upstream boundary of reach N2 in the Nicola River, which was located at the confluence with the Coldwater River (see Section 2.1). The Nicola River from Nicola Lake downstream to the confluence with Coldwater River has moderate to high spawning activity (Parken et al. 2003), but was omitted from the study design due to limited resources.

For all tributary reaches we designated just two habitat types, riffles and runs, with the latter type including all lower gradient habitats including pools, runs, glides, alcoves, etc. Previous studies in Thompson River tributaries suggested that juvenile Chinook densities were greater in runs and pools than in riffles (Tredger 1980b; Sebastian 1982). Therefore, in most cases, we assigned greater sampling effort to the run habitat type, reflecting our expectation of higher abundance and associated variability for these strata.

Riffle and run habitat stratification was inappropriate for the Thompson River mainstem reaches, where wetted width exceeds 100 m and parr are limited to nearshore areas by high midstream current velocities (> 1 m/s). Habitat suitability in these reaches is likely determined by the physical characteristics of the shoreline such as depth, velocity and bank material composition. For the Thompson mainstem, we designated four shoreline habitat type strata that represented the diversity of shoreline types, and which a previous study suggested were likely to reflect variation in juvenile Chinook abundance (Beniston et al. 1985). These were: bars - shallow areas of laminar flow with gravel or cobble substrates that slope gradually away from the bank, deep/fast shorelines - depth greater than 0.5 m and velocity greater than 0.3 m/s within 5 m of the shore, embayments - slack water along the shoreline irrespective of other physical characteristics, and rapids - velocity greater than 1 m/s within 3 m of the shore. Rapids essentially provided no juvenile habitat and were not sampled or included as part of the usable stream length.

Wherever possible, we relied on previous habitat surveys (Tredger 1980b, 1980c; Sebastian 1982) to provide estimates of the total length and area for tributary strata (Deadman, Bonaparte, Spius and Nicola). To obtain habitat data for the remaining tributary (Coldwater) and the Thompson River mainstem, we conducted habitat surveys during October 2002.

Within each stratum, fish populations were sampled using a systematic sampling design, whereby in each mainstem or tributary reach sites were distributed at uniform intervals (with some deviations resulting from access limitations). For the tributary reaches, sampling sites for run habitat strata consisted of entire habitat units. For riffle strata, habitat units were sub-sampled. This was done because: (i) habitat quality in different sections of riffles was less heterogeneous than in runs; and (ii) in most reaches riffles were sampled by single-pass electrofishing, which was too time-consuming to allow for sampling of entire habitat units in all but the smallest streams. For riffles, each site was a minimum of 100 m² in area and 10 m in length. For the three habitat types in the Thompson River mainstem reaches, site length equaled the entire length of the unit or 100 m if the shoreline habitat unit exceeded this length.

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

2.3 Night snorkeling surveys

We used calibrated nighttime snorkeling surveys, adjusted for snorkeling efficiency, as our principal method for fish population sampling, with single-pass electrofishing substituted for snorkeling in riffle strata in small or shallow tributary reaches. Numerous studies have shown that daytime concealment behaviour is common in juvenile salmonids (e.g., Bradford and Higgins 2000 and references therein), and 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 limited the underwater surveys to a four-hour period beginning 0.5 hours after dusk, based on Bradford and Higgins' (2000) finding in the nearby Bridge River that, throughout the year, the highest abundances of Chinook fry were always observed during the 3-4 hours after dusk.

In all streams except the Thompson River mainstem, snorkelers surveyed the stream's entire wetted width. At sampling locations in the Thompson River snorkelers moved out as far as was physically possible from shore or until no fish were observed. To determine if fish were present in the higher velocity mid-channel portion of a Thompson River site where upstream travel was not possible, snorkelers would conduct periodic downstream

sweeps. However, this appeared to be generally unnecessary, as juvenile salmonids typically move closer to shore at night (Bonneau and Scarnecchia 1998). In larger mainstem reaches in our study area, Chinook fry were concentrated at inshore locations of reduced current velocity at night.

To illuminate the sampling sites at night, snorkelers used handheld dive lights. In very clear or small sites, snorkelers frequently diffused the beams by reflecting them off the underside of the stream surface or by employing a sanded filter over the lens to minimize the disturbance to fish. 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. Regular communication between snorkelers was essential to avoid duplicating counts, particularly in the instances where fish were distributed across the entire wetted width of the stream. In the smallest stream reaches (Criss, Upper Spius), and in the single shoreline habitat units in the Thompson River mainstem, surveys were conducted by a single snorkeler.

Size data were not collected at snorkeling sites because high numbers of fry made it impractical for snorkelers to estimate individual fish lengths. We assumed all Chinook observed to be age-0+ fry based on previous adult scale analysis for the lower Thompson River basin that indicated that the vast majority of stream-type lower Thompson River Chinook migrated seaward after one year (data on file, Aging Lab, DFO Pacific Biological Station).

2.4 Mark-recapture estimates of snorkeling efficiency

During 2004-2006, we used a mark-recapture methodology to calibrate snorkeler counts of Chinook fry in primary and secondary tributaries as well as the Thompson River mainstem. One night prior to conducting snorkeler counts as described above, fry were captured throughout the calibration site by a snorkeler using two large aquarium nets affixed to handles of approximately 80 cm in length, then marked and released. The snorkeler captured fish as he encountered them during thorough searches at locations spread systematically through the site, and large or small fish were not selected

preferentially. Minimizing site disturbance was a primary goal of the marking methodology. Each captured fry was handed to a second crew member on shore, who immediately measured it (fork length to nearest 5 mm), tagged it, and returned it to its original lie. Tagging consisted of inserting a size 18, barbed fish hook, attached to a 15 mm long piece of coloured plastic chenille, through the skin of the fish's back at the insertion of the dorsal fin. Captured fry were not anaesthetized because of uncertainty about behavioural effects from the anesthetic, and because they could be measured and marked without being removed from the net. Fish were returned to their lie only after the snorkeler had moved beyond it. During the snorkeling survey the following evening, marked and unmarked fry were recorded separately, so that mark-recapture abundance estimates could be calculated for each of the calibration sites, and snorkeling efficiency 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; Bohlin et al. 1989; Mitro and Zale 2002). We chose to conduct the underwater surveys 24 hours 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. 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. To make surveys less time-consuming 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 re-sighted in a mark-recapture study, we targeted 15-20 marks per site,

with 10 marks considered a minimum. This represented a trade-off between achieving relatively precise estimate of mean site-level snorkeling efficiency (i.e. large number of sites) versus precise estimates of snorkeling efficiency at individual sites (large number of marks per site).

2.5 Single-pass electrofishing

With the exception of riffles in the lower Nicola River reach and riffles in both reaches of the Bonaparte River, tributary riffles were too shallow to be surveyed by snorkelers, and were therefore sampled by a single electrofishing pass, with the catch adjusted for capture efficiency to provide an abundance estimate. In the lower Nicola and Bonaparte rivers, higher flows ($>4 \text{ m}^3/\text{s}$) and a relatively confined stream channel provided sufficient depth for snorkelers to effectively survey riffles instead of electrofishing them. Electrofishing sites in riffles were fully enclosed (one bank to the other) by upstream and downstream stop nets. The exception to this was the relatively wide ($> 20 \text{ m}$) upper Nicola reach, where riffles were sampled at sites along one shore. Here, upstream and downstream stop nets were placed perpendicular to the shore and the offshore side of the site was bounded by water too swift to be utilized by fry. Electrofishing proceeded always from the fast water forming the offshore boundary towards the shore, to avoid chasing fish from the site. At all riffle sites, electrofishing was initiated at the downstream net, and consisted of a thorough search in an upstream direction, followed by a systematic sweep back towards the downstream net.

2.6 Three-pass electrofishing to estimate single-pass and snorkeling efficiency

Maximum likelihood (ML) abundance estimates from three-pass depletion electrofishing were used to estimate capture efficiency for single-pass electrofishing at a subset of sites in riffle strata in smaller tributaries (i.e. Coldwater, Spius, and Deadman). At three-pass sites, the same methodology as that used at single-pass sites was employed, with the first pass treated as the equivalent of the single-pass conducted at non-calibration sites.

For run strata in smaller tributaries, during 2001 and 2002 (prior to initiating the mark-recapture study of snorkeling efficiency) we also conducted three-pass electrofishing at a

subset of the sites (Decker and Hagen 2004), in this case to estimate snorkeling efficiency. These data provide a second calibration method for comparison with the mark-recapture method. At each snorkeling efficiency calibration site, electrofishing was conducted within 24-48 hours following the snorkeling survey. During three-pass electrofishing, run sites were fully enclosed sites with stop nets spanning the entire stream width.

All salmonids captured during single-pass and three-pass electrofishing were anaesthetized, identified to species, measured for fork length (nearest mm), weighed (nearest 0.1 g), and released back into the site after sampling.

2.7 Chinook fry size and stock identification

At most of the sites fish abundance was estimated using snorkeling counts. Size data were not collected at these sites because high numbers of fry made it impractical for divers to estimate individual fish lengths. Length data were collected at sites where electrofishing was used as an alternative sampling method (Section 2.5), at sites where snorkelers captured fish as part of a mark-recapture study to estimate snorkeling efficiency (Section 2.4), or in reaches where electrofishing was conducted to collect genetic samples (see below). As well, in 2002 electrofishing was conducted at several locations in the Bonaparte River to obtain size data for that stream.

To investigate the parental stocks of fry present in the Thompson and Nicola River mainstems, we collected genetic samples (caudal fin clip) during the October field surveys in 2002 and 2006 (2002 only for Nicola River). Genetic samples were collected in the lower two reaches of the Thompson (T2 and T3) and in both study reaches in the Nicola River (Nicola River samples were collected in 2002 only). Samples were obtained from fish captured by electrofishing crews or by snorkelers as part of the mark-recapture study. The number of fry collected at a particular sampling site was limited to 10, so as to reduce the likelihood of collecting a large number of samples produced by the same parents.

2.8 Data analyses

2.8.1 Chinook fry size and stock identification

We estimated mean fry size for each stream in each year for which fork length data were available, and compared fry size among streams and between years using ANOVA and Bonferroni-adjusted pairwise comparisons and hypothesis tests.

Stock composition of fry was estimated by analysis of microsatellite variation at 13 loci using the Fraser River Chinook salmon genetics baseline (Beacham et al. 2006). Stock composition was estimated with a maximum likelihood estimation procedure for multi-locus genotypic frequencies for each population using the Statistical Package for the Analysis of Mixtures software program (J.R. Candy, Pacific Biological Station, DFO, pers. comm.; Debevec et al. 2000). Stock compositions were estimated as the means of repeated runs of the maximum likelihood model, with the sum of the proportions equaling one. Standard deviations of these mean values described uncertainty in the estimates.

2.8.2 Mark-recapture estimates of snorkeling efficiency

Estimates of snorkeling efficiency (q) 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:

$$q_i' = \arcsin \sqrt{q_i} \quad (1)$$

To assess whether normal distribution statistics could be used to model snorkeling efficiency, we performed Kolmogorov-Smirnov one-sample tests of normality (Zar 1996) on both untransformed and transformed data. We then examined whether snorkeling efficiency data from the seven streams should be combined, to improve sample size, or treated separately, using single-factor ANOVA with stream treated as a factor and

estimates of snorkeling efficiency at individual sites as the sample unit. If snorkeling efficiency did not differ significantly among streams, pooled data from all streams were used to estimate mean snorkeling efficiency.

Peterson mark-recapture estimates of abundance will be biased if the key assumption of site closure for marked fish is not met, but increasing the recapture section length in open sites will reduce this bias (Albanese et al. 2003). In estimating snorkeling efficiency at calibration sites, we therefore included marked fish that had moved beyond the original site boundaries in comparing the number of marks seen to the number deployed in the original marking site. We assessed the degree of movement of marked fry by comparing the total number of marked fish, pooled across all sites, that were resighted in the upstream and downstream sections to the number resighted in the original marking sites.

We investigated whether the probability of snorkelers sighting marked fish was related to habitat variables measured at mark-recapture sites using logistic regression, with data pooled for 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 multicollinearity. 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 on the likelihood of marked fish being seen.

Because of evidence of multicollinearity among predictor variables, the significance of individual predictors was evaluated using a second method, as well, using stepwise logistical regression and a backward elimination procedure in which the predictor with the highest P value was eliminated from the analysis and the regression re-calculated, continuing until the P values for all remaining predictors in the regression were statistically significant (Zar 1996).

2.8.3 Single-pass electrofishing capture efficiency and snorkeling efficiency derived from multiple-pass electrofishing

Three-pass depletion electrofishing data were used to compute maximum likelihood (ML) estimates of fish abundance (Otis et al. 1978) at single-pass electrofishing and snorkeling calibration sites in the smaller tributaries. Although the potential for negative bias in the depletion estimates exists (discussed in a later section), depletion estimates were treated as “true” abundance estimates for the purpose of estimating single-pass electrofishing and snorkeling efficiency. To assess whether normal distribution statistics could be used to model single-pass electrofishing and snorkeling efficiency based on electrofishing depletion data, and whether data from all streams could be pooled, we used the same procedures as those described above for the mark-recapture study of snorkeling efficiency.

2.8.4 Fry standing stock estimates

With our sampling design, error in the estimation of mean Chinook salmon fry density and standing stock is the result of both first stage or process error (spatial variation in fish abundance) and second stage or measurement error. Measurement error in the case of snorkeler counts includes variation related to fish behaviour and habitat characteristics among sites, and variation in snorkeling efficiency among crewmembers. With respect to single-pass electrofishing, measurement error includes variation in capture efficiency among sites and among individual fish, as well as differences in efficiency among crews. How to model and combine these error sources when estimating population size for whole streams is controversial. Hankin and Reeves (1988) and Mitro and Zale (2002) presented similar parametric estimators suitable for application to normally-distributed data. However, as is generally the case for stream fish populations, Chinook fry populations in the Thompson River basin are spatially distributed non-normally, with patterns sometimes varying from stream to stream. To address the problem of sparse or non-normally distributed fish density data for many of the strata, we used a non-parametric bootstrap procedure (Efron and Tibshirani 1993) that relies on repeated sampling of the actual data rather than a theoretical probability distribution to quantify uncertainty in population estimates.

For each reach/habitat type stratum i , parr density (D_i) and standing stock (SS_i) were computed simply as point estimates from the sample, since point estimates generally perform better than bootstrap parameters (median or mean) in simulation exercises (Efron and Tibshirani 1993):

$$D_i = \frac{\sum_{j=1}^n N_{ij}}{\beta_i \sum_{j=1}^n L_{ij}} \quad (2)$$

$$SS_i = D_i \times L_i \quad (3)$$

where:

j = sample site in stratum i

n_i = the number of sites in stratum i for which data were collected

N_{ij} = unadjusted snorkeler count or single-pass electrofishing catch at one of n sample sites randomly selected by the bootstrap model from stratum i .

β_i = mean snorkeling efficiency or single-pass electrofishing capture efficiency for stratum i

L_{ij} = stream length (m) for site j in stratum i

L_i = total stream length (m) for stratum i

We then obtained 99% confidence intervals for D_i and SS_i by bootstrapping the data. Bootstrap iterations were computed using R, an open-source software environment for statistical computing and graphics (Version 2.2.1, available at <http://www.r-project.org>). For each reach/habitat type strata, 5000 iterations of equations (2) and (3) were computed with the bootstrap model choosing sample sites with equal probability and with replacement. For each iteration, the number of sites selected was set equal to the actual number of sites sampled. Upper and lower 95% confidence limits for D_i and SS_i were estimated as the 97.5% and 0.025% percentiles, respectively, from the cumulative distribution of the 5000 bootstrap iterations (Efron and Tibshirani 1993). In cases where the median of the bootstrap estimates differed from the point estimate, we used the

difference between the estimates to develop bias-corrected confidence limits (Haddon 2001).

To account for additional measurement error associated with uncertainty in snorkeling efficiency or single-pass electrofishing efficiency we incorporated a Monte Carlo procedure within the bootstrap routine as follows: each sample of fish density selected by the bootstrap algorithm for a particular reach/habitat type stratum was adjusted by stochastically modifying β_{ij} in equation (2) based on the mean and standard deviation computed from the sample data for β_{ij} . In our study, the normal distribution was appropriate for modeling variation in capture efficiency.

To compute standing stocks for each stream and for the lower Thompson River basin as a whole, estimates for the appropriate reach/habitat strata were summed during each bootstrap iteration, and the aggregate standing stock and the upper and lower confidence limit estimates were computed in a manner identical to that used for computing individual strata values. To describe the precision of the standing stock estimates, throughout the report we use percent relative error, which can be defined as the average half confidence interval (UCI minus LCI divided by two) divided by the mean and expressed as a percentage (Krebs 1999). It should be noted that with bootstrapping, the confidence interval is often not symmetrical about the mean.

2.8.5 Physical habitat predictors of Chinook fry abundance

We utilized direct, multiple linear regression (Tabachnick and Fidell 2001) to evaluate whether habitat-based variables were useful predictors of Chinook fry abundance at the site level in the lower Thompson River basin. For the dependent variable, we used mean fish density (numbers/km) across years because estimates of fish abundance from different years at the same site were not independent with respect to their relationship to habitat variables. Site-level physical habitat variables were natural log-transformed to meet requirements for multivariate normality and linearity (Tabachnick and Fidell 2001), as was the dependent variable. Two additional habitat variables were derived from a GIS database: distance from the mouth of the stream and proximate stream gradient using

methods described by Parken et al. (2002) and Williams et al. (1999). For the latter variable, average stream gradient was estimated for 1 km long sections within each stream using GIS streambank elevation data, and individual sites were assigned the gradient estimate for the section in which they occurred. Neither of the last two variables required log-transformation.

To identify multicollinearity, and to increase the power of the regression analysis to evaluate individual predictors, we examined a Pearson product-moment correlation matrix that included all potential independent variable combinations. We set a target level for bivariate correlations among predictors of approximately 0.5 or less, a conservative but appropriate standard given that intercorrelation among fish habitat variables can obscure the importance of the individual variables in multiple regression. Prior to logistic regression analysis, single predictors were selected for inclusion in the model from among groups of redundant and/or highly correlated variables. Among correlated predictors, priority for inclusion in the model was given to those with logical potential for a causal relationship with fry abundance, low correlation with other model-component variables, and a straightforward and accurate means of measurement in the field. We also inspected individual pair-wise correlations of deleted variables with the dependent variable, to confirm that we were not unwittingly dropping a key predictor.

Individual sites were placed in one of seven stream and one of five habitat type categories through the use of two dummy-coded categorical variables. Squared semi-partial correlations (sr_i^2) were used to assess the unique contribution of statistically significant predictors to the simple multiple linear regression.

A question of particular interest to us was whether Chinook fry abundance was more variable in ‘marginal’ relative to ‘core’ rearing habitats, suggestive of a pattern of habitat use where the greatest differences in fish density among high and low escapement years occur in marginal habitats, with fish densities in high quality habitats remaining relatively static. This question has important implications for designing stock-recruitment studies that are sensitive to modest changes in juvenile abundance. We evaluated year-to-year

variation in Chinook fry abundance using a second multiple linear regression, with the coefficient of variation (CV), or standard error divided by the mean, as the dependent variable, and independent variables delineating core and marginal habitats selected on the basis of the previous multiple regression. Coefficients of variation were calculated only for sites for which at least three years' abundance estimates existed, and for which at least two years' estimates were not zero.

We also evaluated whether interspecific competition between Chinook fry and juvenile steelhead was occurring, using correlation analyses for steelhead fry and age-1+ parr abundance (Hagen et al. 2007 in prep.) versus Chinook fry abundance among years. Stream-specific estimates of total abundance for a given year are not independent, and therefore we compared standing stocks for the total study area only. Hagen et al. (2007 in prep.) did not calibrate snorkeling counts for steelhead fry, therefore reliable abundance estimates for steelhead fry were limited to shallow riffle habitats where calibrated single-pass electrofishing was substituted as the sampling method. In testing for correlations between Chinook and steelhead fry standing stocks, we used only data from shallow riffle strata.

2.8.6 Stock-recruitment analyses

Developing a reasonably reliable stock-recruitment relationship usually requires 15 or more years of data (Parken et al. 2006). To supplement the six years of data from our study, we searched the literature for previous estimates of fry density or standing stock for lower Thompson River tributaries². To be acceptable for presentation in comparison with results from this study, estimates of fry abundance from other studies had to meet several criteria: 1) sampling must have been conducted with the intent of estimating mean fry density or total standing stock for an entire stream as opposed to specific reaches or habitat types; 2) a minimum of 10 sites had to be sampled in each stream; and 3) fish

² The lower Thompson River mainstem was excluded because this stock is thought to be exclusively ocean-type (Parken et al. 2002), thus, the fry population present in the Thompson River mainstem in October likely consists of fry from the lower Thompson tributaries and spawning areas upstream of Kamloops Lake.

abundance at each site must have been estimated using a reliable quantitative sampling method (e.g., multiple-pass removal, mark-recapture, etc.). All escapement data was obtained from DFO (DFO, SEDS database). Although the lower reaches of the Coldwater River and Spius Creek are seeded by the spring-run Nicola stock rather than by distinct early spring-run stocks that spawn in the upper reaches of each stream (see Section 2.3), we combined fry and spawner numbers for the upper and lower reaches because pre-1986 fry and escapement estimates for Coldwater River and Spius Creek were not reach-stratified. Therefore, values for the Nicola River excluded fry and spawner numbers in the lower reaches of these streams. We did not sample fry numbers in the section of the Nicola River between Nicola Lake and the Coldwater River confluence³ in our estimates of standing stock for Nicola River (Figure 1), but we did include spawner numbers in this section as part of the total for Nicola River because, again, pre-1986 escapements estimates were not reach-specific.

Detailed stock-recruitment analyses, including the estimation of spawning stock reference points based on S_{msy} (spawning stock size associated with maximum sustainable yield; CTC 1999), and potential correction for bias in estimates, were beyond the scope of this report given the limited stock-recruitment data available. Instead, we focused on using the available data to examine whether relationships existed between brood spawner abundance and fry standing stock for individual tributary stocks and aggregate stocks (e.g., Nicola River basin) over the range in brood escapements represented by our fry abundance data set.

To help delineate the form adult-to-fry stock-recruitment relationships for tributaries streams, we fit both Ricker and Beverton-Holt curves (Ricker 1975) to the data. A commonly used version of the Ricker model is

$$R = aSe^{-bS} e^c \quad (4)$$

³ Earlier studies suggested that fry numbers in this section did not contribute significantly to total abundance in the Nicola River (Sebastian 1982).

where R is recruitment resulting from S , the spawning stock, a is the number of recruits per spawner at low stock size, b relates to the rate of decrease of recruits per spawner as the spawning stock increases, and e^ε indicates that the residual errors between predictions of R and the observed production of recruits are expected to be log-normal (Haddon 2001). In order to fit the Ricker stock-recruitment model with lognormal residual errors to our data, we used the log-transformed equation (Haddon 2001)

$$\text{Ln}\left(\frac{R}{S}\right) = \text{Ln}(a) - bS + \varepsilon \quad (5)$$

to normalize the error structure. We then fit the equation using ordinary least squares to determine optimal values of a and b for each set of spawner-to-recruit data.

The Beverton-Holt stock-recruitment model is asymptotic in nature, with the maximum recruitment asymptote reflecting the carrying capacity of the environment for a particular cohort. In the form

$$R = \frac{aS}{b+S} e^\varepsilon \quad (6)$$

R , S , and e^ε are recruitment, spawning stock size, and lognormal expectation for residual errors as above, but a and b have quite different meanings; a is the maximum number of recruits produced, and b is the spawning stock needed to produce recruitment equal to half the maximum (Haddon 2001). We used the log-transformed version

$$\text{Ln}\left(\frac{R}{S}\right) = \text{Ln}(a) - \text{Ln}(b+S) + \varepsilon \quad (7)$$

to find optimal values of a and b for each dataset using the method of ordinary least squares (Haddon 2001).

In a separate analysis, we assessed the relationship between spawner abundance and resultant fry abundance using pooled data for all streams in all years. To facilitate comparisons of stock-recruitment relationships among streams that varied in total length and productive capacity, we standardized the data by transforming abundance estimates to densities (spawners/km and fry/km). To distinguish between spawner density effects and possible stream and year effects, we used ANCOVA with stream and year as factors (separately) and spawner density as the covariate.

Finally, we investigated whether fry abundance data were useful for predicting future adult returns. Lower Thompson River stream-type Chinook predominately spend two years in the ocean (e.g., 2003 fall fry smolt in 2004 and return as adults in 2006). Escapements estimates were only available for the 2001-2003 fry years included in this study, and for previous fry years when surveys when intensive surveys were conducted as part of other studies (see above). Given the paucity of data, we limited this analysis to simple plots of spawner density versus brood fry abundance for individual streams, without attempting to fit stock-recruitment curves to the data.

3.0 RESULTS

3.1 Fry sampling

From 2001 to 2006, we sampled Chinook fry abundance at 110-146 sites that were distributed throughout the major portion of the lower Thompson River basin utilized by adult spawners (Table 2; Figure 1). Of these sites, 96-120 sites were sampled by snorkeling, and the remainder, which represent shallow riffle strata in smaller tributaries, were sampled by single-pass electrofishing. An average of 6,451 linear m of stream channel was sampled each year, which equated to a sampling intensity of 1.4%. In tributary reaches, site length ranged from 12-138 m for run habitat strata, and 10-28 m for riffle strata. For the Thompson River mainstem reaches, length of the single-shore sites ranged from 50-150 m. The survey was completed during October each year, and, in most cases, sampling conditions (streamflow and turbidity) remained good to excellent. One exception was 2005 when greater than normal rainfall caused streamflow in Spius

Creek and Coldwater River to remain well above 1 m³/s for most of October. As a result we were unable to use electrofishing to sample shallow riffles in these streams, and these strata went unsampled in 2005 (Table 2). Similarly, higher than average flows in Bonaparte River increased the silt load and prevented us from conducting snorkeling counts. No abundance data were collected for Bonaparte River in 2005.

October water temperatures ranged from 5°C to 13°C in Coldwater and Deadman Rivers and lower Spius Creek, from 6°C to 9°C in lower Bonaparte River, from 5°C to 10°C in the upper reaches of Spius Creek and Bonaparte River, and from 7°C to 12°C in Nicola and Thompson Rivers.

3.2 Mark-recapture calibration of snorkeling counts

Mark-recapture data were obtained from a total of 40 snorkeling sites distributed throughout all seven study streams in the lower Thompson basin between 2004 and 2006. The number of Chinook fry marked at each site varied from 2 to 26 (mean = 16 marks per site), and our target of at least 10 marks per site was met for all but four sites (Appendix 1).

Snorkel counts were adequate predictors of Peterson mark-recapture estimates at calibration sites ($R^2 = 0.66$), suggesting that in the Thompson River basin they provide reasonably reliable indices of Chinook fry abundance. Two outliers associated with low snorkeling efficiency were identified during regression analysis (*n10* in the Nicola River in 2006 and *tGR* in the Thompson River in 2005; Figure 2a, Appendix 1), but these were kept in the analysis because we assumed they represented real variability. The normal distribution appeared appropriate for modeling variation in snorkeling efficiency among sites (Figure 2b), with arcsine square root-transformed data being a worse fit to expectations for the normal distribution than the untransformed data (transformed data: Kolmogorov-Smirnov $P = 0.79$; raw data: $P = 0.36$). Therefore, we conducted further analyses on untransformed data. Mean snorkeling efficiency (proportions were calculated as counts divided by Peterson mark-recapture estimates) across the 40 mark-recapture sites was 53% (95% confidence interval: 48%-59%, Table 3), and while

differences among streams existed (range: 46% in Coldwater River to 67% in Deadman), they were not statistically significant (ANOVA: $F = 0.99$, $P = 0.45$).

The assumption of site closure did not appear to be greatly violated at mark-recapture sites. During diver surveys one night after marking had occurred, relatively high proportions of the number of marked fry released into the sites were seen (53% on average over all sites as mentioned above), and few fish had moved from the marking site to adjacent upstream and downstream sections (total stream length surveyed for marks was twice the length of the marking site). In total, only 13 of the 330 marked fish resighted by snorkelers at the 40 sites had moved from the original site to one of the adjacent sections (Appendix 1). On average, 2.1% and 1.8% of the total number of marked fish observed during the recapture swims were found in the upstream and downstream sections, respectively.

Intercorrelation among habitat variables measured at mark-recapture sites was high, so several were eliminated prior to conducting logistic regression to evaluate factors affecting the likelihood of individual fish being seen. This was done to avoid multicollinearity and improve model simplicity and power. Variables eliminated included D50, percent boulder, percent cobble and percent gravel substrates, site length, mean thalweg depth, substrate cover, large wood debris cover, deep water cover, undercut bank cover, and overhead cover. Pairwise correlations among the remaining variables in the model (D90, maximum site depth, site width, site area, and turbulence cover) had r values less than 0.7, as recommended by Tabachnick and Fidell (2001). The D50, percent boulder, percent cobble, percent gravel, and substrate cover variables were correlated with the two substrate variables that were retained, D90 and percent fine substrates. Similarly, mean thalweg depth and deep water cover were highly correlated with maximum site depth. Large wood debris, undercut bank, and overhead cover variables were eliminated from the analysis because all were rare; most reaches in the study area had broad alluvial channels, and undercut banks and large wood debris were isolated from the wetted channel during the low flow period in October.

Snorkeling efficiency was not greatly influenced by differences in habitat or other factors among streams. Direct logistic regression with the probability of marked fish being seen as the dependent variable and D90, maximum site depth, site width, site area, and turbulence cover as the set of predictors was statistically significant ($P = 0.02$, McFadden's *rho-squared* = 0.02, $n = 635$), although the only significant individual predictor in the regression was percent fine substrates, which was positively related to the likelihood that marked were seen (t -ratio = 3.60, $P < 0.001$). The value of McFadden's *rho-squared* for the logistic regression, a measure analogous to R^2 in linear regression, indicated that the amount of variance in the likelihood of marked individuals being seen that was explained by the predictors as a group was very low (2% shared variance). Prediction success, as measured by the regression equations ability to predict the original data, was similarly low (51.4%).

Underwater visibility data existed only for approximately half of the observations of individual marked fish. To evaluate the importance of this predictor in logistic regression, we conducted a second analysis on this reduced data set, in which all the predictors of the previous logistic regression were included, but interpretation of statistical significance was restricted to the visibility variable. Horizontal underwater visibility in this analysis was not a significant predictor of the likelihood of individual marked fish being seen (t -ratio = -1.220, $P = 0.22$).

We used linear regression to examine whether snorkeling efficiency could be modeled more precisely by incorporating percent fine substrate as a predictor variable. The result was consistent with that from the logistic regression; percent fine substrates was a marginally significant predictor ($F = 4.00$, $P = 0.05$, $n = 40$), accounting for only 9.5% of the variation in snorkeling efficiency among sites. These results indicated that the simplest approach to modeling snorkeling efficiency at sampling sites, using the mean value of 53% across all sites, with the standard deviation of the mean as the error term, was the most appropriate.

3.3 Three-pass electrofishing calibration of single-pass electrofishing catches and snorkeling counts

Three-pass depletion estimates were used to estimate single-pass electrofishing capture efficiency at 11 tributary riffle sites (Appendix 2). Single-pass catches were excellent predictors of three-pass depletion estimates of Chinook fry abundance ($R^2 = 0.97$, Table 3, Figure 2c). Despite the limited number of calibration sites, it appeared that the normal distribution was also suitable for modeling single-pass efficiency in tributary riffles (Kolmogorov-Smirnov test, $P = 0.75$). Single-pass electrofishing capture efficiency in tributary riffles was high, averaging 78%, but the uncertainty in this value (95% CI for mean: 63-93%; Table 3) was higher than the uncertainty in the mark-recapture derived estimate of mean efficiency for strata sampled by snorkeling. The low number of calibration sites for single-pass electrofishing and the low abundance of Chinook in those sites (Appendix 2) likely contributed to uncertainty. Low sample size precluded an analysis of whether habitat variables were useful predictors of single-pass capture efficiency.

In addition to mark-recapture calibrated of snorkeling counts during 2003-2006, three-pass electrofishing calibration of snorkeling counts was conducted at 20 sites representing tributary run strata in 2001-2002 (Appendix 2). Predictions of snorkeling efficiency were more uncertain than those derived from mark-recapture. Two obvious outliers were removed from the analysis, leaving 18 sites for calibration. The first outlier site (site *spD*, Appendix 2) had a snorkeling count of zero and a three-pass ML estimate of one fry, which fails to provide a meaningful estimate of snorkeling efficiency. In the second case (site *d10*, Appendix 2), the snorkeling count was 11-fold higher than the ML estimate (11 versus 1), indicating severe negative bias in the ML estimate.

Snorkeling efficiency estimates based on depletion electrofishing were not significantly different from expectations for the normal distribution (Kolmogorov-Smirnov, $P = 0.56$) after the removal of the two outliers discussed above, and were used to compute an average snorkeling efficiency estimate of 70% (Table 3). Inter-site variation in depletion electrofishing-derived estimates of snorkeling efficiency was high.

The 95% confidence interval for mean snorkeling efficiency based on electrofishing calibration (47-92%) was much less precise than that for the snorkeling efficiency estimate derived from mark-recapture (48-59%). Despite this, snorkeling counts explained similar proportions of the variation in three-pass electrofishing ($R^2 = 66\%$, Figure 2d) and mark-recapture estimates ($R^2 = 66\%$, Figure 2a), which may reflect the influence of the outliers discussed above for the latter relationship. Given the lower precision of the predictive relationship between electrofishing depletion estimates and snorkeling counts, and, more importantly, the fact that electrofishing calibration sites were limited to small tributaries that could be spanned with stop nets, we used only the mark-recapture-based snorkeling efficiency model to estimate Chinook fry abundance.

3.4 Fry length

Length data was not available for all streams in all years. With the exception of Bonaparte River for which only two years of data were available, each stream had 4 to 6 years of data (Figure 3). There was a significant ($P < 0.001$) interaction between year and stream. Therefore, fry mean fork length was compared among streams for individual years, and among years for individual streams. Fry length differed significantly among streams in every year (Bonferroni-adjusted P values < 0.05 in each case), with the Thompson, Deadman and Bonaparte Rivers having larger fish than the Nicola and its two tributaries, Coldwater River and Spius Creek (Figure 3). Coldwater River tended to have the smallest fry of any stream. In 2005 and 2006, fry in Thompson River were considerably larger than those in the other streams. Fry size also differed significant among years in every stream (Bonferroni-adjusted P values < 0.05 in each case). Fry were smallest in every stream in 2003. Fry were largest in 2004 or 2005, with the exception of Deadman River, where fry were largest in 2001.

Although data were limited, we used ANCOVA to test whether year-to-year variation in mean fry length in tributary streams was related to fish density. Bonaparte River was excluded from this analysis due to limited data. The lower Thompson River mainstem was also excluded because size and numbers of fry there is dependent upon immigration from multiple sources (lower and north Thompson streams and Kamloops Lake). The

interaction between stream and fry density (fry/km) was non-significant ($P = 0.69$), and was removed from the ANCOVA model. With the interaction term removed, mean annual estimates of fry length was negatively associated with fish density (ANCOVA, $F = 6.42$, $P < 0.02$, $df = 1$, $n = 21$; Figure 4), implying density-dependent growth at densities observed during 2001-2006. ANCOVA results suggested a possible relationship between fry length and stream as well ($F = 2.83$, $P = 0.07$, $df = 3$), with mean length being greater at a given fry density in Deadman and Nicola Rivers compared to Coldwater River and Spius Creek (Figure 4). There was no obvious negative relationship between mean fry size and density in the Thompson River mainstem (data not shown), but there were only four years of data available, making this result highly uncertain.

3.5 Stock identification of fry in the Thompson and Nicola rivers

Genetic stock identification results were inconclusive for fry samples collected in the Nicola River basin in 2002. Small baseline sample sizes (Beacham et al. 2006) and low genetic distinctiveness for Spius, Coldwater, and Nicola, (Bailey et al. 2001; Beacham et al. 2003a, 2003b) and stock mixing in early hatchery practices may all have contributed to low precision the estimates. Of the samples collected in the lower Thompson River, a substantial proportion were identified as Deadman River fish (Table 4), but none were identified as being from the Bonaparte River, despite the presence of a larger spawning population in the latter stream. Bonaparte and Deadman rivers are close and fish externally marked at the Bonaparte River fishway have been recaptured later at the Deadman River fence. It appears that at least some adult fish migrate between these two systems and potential matings of Deadman and Bonaparte origin fish could reduce the genetic distinctiveness of fish originating from each river. Previous work also suggests that these two stocks are less accurately identified using genetic variation in 13 microsatellite markers compared to other Fraser River Chinook populations (C. Parken, pers. comm.). We assumed that fish identified as being from Deadman River could have been from either stream.

During October 2002, the majority of the Chinook fry collected from the middle (T2) and lower canyon reaches (T3) of the Thompson River (76% and 68%, respectively;

Table 4) were the progeny of populations from the North Thompson River basin upstream of Kamloops lake, with the primary contributors being stocks from the North Thompson River mainstem and the Clearwater and Barrier rivers (45%, 9% and 13%, respectively). The remaining fry were the progeny of the Deadman and Bonaparte Rivers (14% in T2 and 6% in T3) and the Nicola River basin (combined totals for the Nicola, Coldwater and Spius: 10% in T2 and 27% in T3).

During October 2006, North Thompson Chinook represented a considerably lower proportion of the fry collected from the middle and lower reaches of the Thompson River (39% and 52%, respectively; Table 4); the North Thompson River mainstem, and the Clearwater and Barrier Rivers were again the major contributors for the North Thompson (24%, 11% and 9%, respectively). Fry from the Deadman and Bonaparte Rivers represented 57% and 13% of samples from the middle and lower reaches of the Thompson River in 2006, which was four- and two-fold higher, respectively, compared to values for 2002. In 2006, 33% of the fry sampled from the lower canyon reach of the Thompson River originated from the Nicola River basin. The 2002 and 2006 samples did not include any progeny from the lower Thompson River mainstem stock, and less than 1% was estimated to have originated from the South Thompson basin.

3.6 Fry abundance

Chinook fry were present in all reaches within the study area during 2001-2006, but their distribution was highly non-uniform. Across the 12 reaches in the study area, mean density (fry/km) of Chinook fry varied by as much as two orders of magnitude in some years (extreme values = 40 and 9600 fry/km; Table 5). Despite this variability, there was a consistent pattern of increasing fry densities from upstream to downstream reaches, both within individual streams, and within the study area as a whole. In streams where we delineated more than one reach, fry densities in the downstream reach (average across years) were three to four-fold higher than those in the upstream reach (reach T3 versus T1 in Thompson River). Across the study area, we observed the highest average fry density (5600 fry/km) in the lowermost reach of the study area (reach T3 in the Thompson River mainstem), and progressively lower fry densities in upstream reaches

including those in the Thompson River mainstem, the Nicola River and its tributaries and the Deadman and Bonaparte Rivers. In reaches furthest upstream in the study area (upper reaches of Coldwater and Bonaparte Rivers and Spius Creek), average densities ranged from 540 to 830 fry/km, or about 7-10-fold lower than at the downstream end of the study area.

Among tributary reaches, Chinook fry density in runs was, on average, 9.9 times greater than in riffles (range = 0.9 to 127.4 times greater in runs; Appendix 3a-b). Differences in density among the three habitat types in the Thompson River mainstem were less consistent, but Chinook were generally most abundant in embayments, followed by cobble bars, with deep-fast habitats supporting the lowest fry densities (Appendix 3a-b).

In order to obtain estimates of total standing stock for the lower Thompson basin and for individual streams in 2005, in the absence of abundance data for Bonaparte River and in riffle strata in Coldwater River and lower Spius Creek, we approximated standing stock values for missing strata. We computed a fry standing stock estimate for Bonaparte River in 2005 by summing the 2005 standing stocks for the other streams and factoring this value by the average proportion of the overall standing stock contributed by Bonaparte River in other years of the study. For riffle strata in Coldwater River and Spius Creek in 2005, we estimated standing stocks by factoring 2005 standing stocks in run strata by the average ratio of riffle to run standing stock in other years of the study. Confidence intervals for 2005 standing stock estimates for these streams and the Thompson River basin as a whole do not take into account the error in these approximations.

During 2001-2006, the estimated total standing stock of Chinook fry in the lower Thompson River basin varied three-fold from 480,000 fry in 2004 to 1.5 million fry in 2003 (Table 5; Figure 5a). Covariation among streams in year-to-year fry abundance was evident, but adjacent streams were not necessarily more strongly correlated than more distant ones (Table 6). . In the tributaries, fry standing stocks were two to nine-fold

lower in 2004 compared to other years (Figure 5). In the Thompson River mainstem fry abundance was lowest in 2006. Despite the high inter-annual variation in fry abundance, with the exception of 2004, the proportion of the total standing stock for the study area contributed by each stream was very stable from year to year (Figure 6).

3.7 Physical habitat predictors of Chinook fry abundance

The seven continuous and two categorical habitat variables tested as predictors of mean fry abundance at individual sites were: 1) log-transformed D90 (lnD90), and 2) percent fine substrates (lnFINESUB), which were the best combination representing correlated substrate variables; 3) mean thalweg depth (lnDEPTH), which represented depth and depth-related cover variables; 4) percent turbulence cover (lnTURB); 5) volume of large wood debris (m³) (lnLWD); 6) distance from the stream mouth (DISTANCE); 7) proximate stream gradient (GRADIENT); and the categorical variables (8) stream (STREAM) and (9) habitat type (HABTYPE; see Table 2). The cover variables percent overhead cover and percent undercut banks were dropped as lower priority because such cover is relatively rare within the wetted channel of lower Thompson basin streams during the fall, and the very high proportion of zero observations made data difficult to transform to meet normality requirements.

After the deletion of 33 cases because of missing values, mean Chinook fry abundance data were available for 210 sites. This number was higher than the number of sites sampled annually (section 3.1) because many sites were not sampled in all years. The habitat variables as a set were highly significant predictors of mean fry abundance ($F = 22.4$; $P < 0.001$), explaining 58.5% of among-site variation (Table 7). The regression analysis indicated that variation in fish density among sites was driven by distance from the stream mouth (negative, $F = 56.0$, $P < 0.001$), habitat type category ($F = 18.3$, $P < 0.001$), and stream ($F = 4.04$, $P = 0.002$), the same factors that influenced fry density among reaches (see previous section). Stream gradient (negative, $F = 23.9$, $P < 0.001$) and a lone predictor derived from site-level habitat measurements, mean site depth (positive, $F = 5.59$, $P = 0.019$) were also statistically significant. The order of importance for the regression solution (based on squared semi-partial correlation) was

DISTANCE ($sr_i^2 = 0.118$), GRADIENT ($sr_i^2 = 0.050$), HABTYPE ($sr_i^2 = 0.039$), lnDEPTH ($sr_i^2 = 0.012$), and STREAM ($sr_i^2 = 0.009$). The remaining site-level habitat variables lnD90 ($F = 1.69$, $P = 0.196$, $sr_i^2 = 0.004$), lnFINESUB ($F = 0.493$, $P = 0.483$, $sr_i^2 = 0.001$), lnTURB ($F = 0.162$, $P = 0.688$, $sr_i^2 < 0.001$), and lnLWD ($F = 0.050$, $P = 0.823$, $sr_i^2 < 0.001$) were not significant predictors in the regression solution.

To test whether annual variation in fry density was more or less variable in marginal relative to core rearing habitats, we performed direct, multiple linear regression of the coefficients of variation for fry density estimates among years ($CV_{\text{fry density}}$), for sites with multiple years' data, on the three most important predictors of mean site abundance: distance from the mouth, gradient, and habitat type. We assumed that sites representing marginal habitat occurred in riffle habitats, in high gradient stream sections areas, and in areas distant from the stream mouth, while sites representing core rearing habitat occurred in runs, lower gradient sections areas, and areas near the stream mouth. Thompson River mainstem sites were not included in this analysis because a high proportion of Chinook fry found there did not originate in the lower Thompson River basin, and because we were uncertain as to the nature of colonization of these habitats by lower Thompson fry.

After deletion of cases with missing abundance data, coefficients of variation for 103 sites were available for the analysis. The predictors as a set reliably predicted $CV_{\text{fry density}}$ at individual sites ($F = 10.9$, $P = 0.001$), although the amount of explained variation was relatively low ($R^2 = 13.2\%$), and the only significant individual predictor was habitat type category ($F = 11.870$, $P = 0.001$, $sr_i^2 = 0.102$). Chinook fry density in riffles was more variable among years than was fry density in runs, consistent with the notion that run habitats are utilized more consistently in addition to being more productive. Although not significant predictors in the regression solution, distance from the stream mouth ($F = 3.143$, $P = 0.079$, $sr_i^2 = 0.027$) and stream gradient ($F = 1.686$, $P = 0.197$, $sr_i^2 = 0.014$) were both positively correlated with $CV_{\text{fry density}}$ ($r = 0.11$ and 0.09 , respectively), also consistent with the idea that abundance in lower quality habitats is more variable from year to year.

We also considered the potential for interference competition from juvenile steelhead as a factor affecting Chinook fry abundances. Because mean abundances of age-1+ steelhead and Chinook fry were positively correlated at sites we assumed that habitat quality differences among sites would obscure any possible effects of competition, and therefore we did not include steelhead parr abundance in the multiple regression analysis. Instead, we assessed interspecific competition by evaluating the correlation between total Chinook fry and total age-1 steelhead parr abundances for the lower Thompson system as a whole. Steelhead parr standing stock was positively correlated with Chinook fry standing stock during the same year ($r = 0.59$), but was negatively correlated with Chinook fry standing stock from the previous following year ($r = -0.27$), suggesting that competition with Chinook salmon negatively impacts juvenile steelhead abundance primarily when they are fry. Neither correlation was significant ($P = 0.21$ and 0.65 , respectively), but sample size was very small ($n = 6$ years). The idea that interference competition was occurring between Chinook fry and steelhead fry was not supported, however, by a more direct comparison of their abundances. Yearly standing stock estimates for the total of all shallow riffle strata sampled by electrofishing, the only strata for which steelhead fry abundance estimates can be considered reliable, were strongly positively correlated ($r = 0.85$; $P = 0.034$).

3.8 Fry abundance versus spawner abundance

In general, robust methods such as direct counts at fishways, fences and resistivity counters, or mark-recapture methods were used to estimate escapement after 1990 whereas less intensive methods (expanded peak counts from aerial surveys) were used during earlier years and at Spius Creek and Coldwater River (Table 8). Although these are the best indices of spawner abundance available, spawner escapement estimates prior to 1990 are less reliable than those after 1990 and the accuracy of estimates should be interpreted cautiously. With the exception of Bonaparte River the range in escapement for each tributary was relatively broad considering the limited amount of data available (6-9 years depending on the stream, Table 8). With the exception of the Deadman River,

brood spawner and fry abundance in lower Thompson River tributaries was generally higher during this study (2001-2006, Table 8) relative to previous years (1978-1991, Table 8). No fry data from previous years were available for Bonaparte River. For individual tributaries, the largest brood escapement was 6- to 48-fold greater than the smallest escapement, with the exception of Bonaparte River where the range was only 2.4-fold.

Chinook fry standing stock was positively related to brood spawner escapement (Figure 7), when fry standing stock data for 2004 were excluded (2003 escapement; open diamonds in Figure 7). The among-year contrast in escapement for Bonaparte River was insufficient to determine whether fry abundance was positively related to spawner abundance. In all study tributaries, fry density in 2004 was uniformly low (Figure 5), despite strong escapements in 2003 relative to other years (Figure 7), indicating the confounding influence of a density independent mortality factor(s) that can strongly affect fry production. Apparent low egg-to-fry survival in 2004 coincided with severe low streamflow/high temperature conditions in 2003 during the final migration and spawning period for the brood spawner cohort (WSC streamflow data on file, MOE and DFO water temperature data on file). In the case of Coldwater River, a second, clear outlier was low fry abundance in 1981 (Figure 7). In December of 1980 a major rain-on-snow event occurred in the Nicola River basin, which increased flows in Coldwater River from $\approx 2 \text{ m}^3/\text{s}$ to $\approx 95 \text{ m}^3/\text{s}$ (WSC, station 08LG048), and likely resulted in the formation and breakup of ice dams subsequently causing streambed scour. Eggs incubating in the Coldwater River likely suffered significant mortality because fry were virtually absent by the fall.

In fitting Ricker and Beverton-Holt stock-recruitment curves (Figure 7, Table 9), we excluded 2004 data for all streams and 1981 data for Coldwater River. For some uses of the data this may be inappropriate (CTC 1999). In our case, however, we were interested only in density dependent survival of Chinook fry in the lower Thompson basin, and in investigating the form of the density-dependent relationship. Plots of Chinook adult-to-fry stock-recruitment for lower Thompson River tributaries suggest density dependent survival for Spius Creek, Coldwater River, and possibly for Bonaparte and Nicola Rivers

as well (Figure 7). However, spawning stock sizes during the study were not high enough and did not exhibit sufficient contrast to evaluate whether data were best represented by a Beverton-Holt, Ricker, or some other recruitment relationship at high spawner abundance. This was particularly true for the stocks in the Deadman and Nicola rivers, where fry abundance appeared linearly related to spawner abundance over the range of stock sizes observed. Beverton-Holt and Ricker recruitment curves look very similar for these two populations (Figure 7).

For the sake of comparison, we included predictions of S_{msy} from the watershed area-based model of Parken et al. (2006) in the plots of fry versus spawner abundance for individual tributaries (Figure 7), with S_{msy} values depicted as vertical dotted lines. For Coldwater, Bonaparte and Nicola Rivers, and for the Nicola basin as a whole, the inflection point of the Ricker curve derived from our data occurs at spawner abundances near, but somewhat less than the S_{msy} estimates from Parken et al. (2006). For Spius Creek, our data suggests a more substantial overestimate of S_{msy} by the watershed area model, consistent with the findings of Parken et al. (2006). For Deadman River, spawner abundances during the study appeared well below S_{msy} , since there was no apparent inflection in the Ricker curve, precluding an evaluation of the watershed area-based estimate of S_{msy} (2,400 spawners; 80% confidence interval: 1,900-2,800 spawners). A comprehensive evaluation of the habitat model predictions was limited by the range of spawner abundances, little data, and potentially variable negative bias in estimated fry production due to fish emigrating from natal streams to rear in the lower Thompson River and areas downstream, and difficulties discerning fry production from the early and late stock components in the Nicola watershed. Further, estimates of S_{msy} from fry production data and the habitat model are both imprecise, with coefficients of variation of 15-30% for S_{msy} (Parken et al. 2006). Additional fry production estimates from higher spawner abundances are needed to increase confidence in the spawner-fry production relationships.

When spawner and fry densities from individual tributaries were plotted together (excluding outliers), Nicola and Bonaparte Rivers had the highest maximum values for mean annual fry density (fish/km, Figure 8, upper graph), and Deadman River and Spius

Creek had the lowest. Bonaparte and Nicola rivers also had the highest values for maximum spawner density (Figure 8, upper graph). The range in spawner density for which density dependent decreases in fry survival were apparent varied considerably among tributaries. For example, in Nicola and Bonaparte Rivers fry abundance continued to increase with spawner density well beyond the levels of density dependent survival for Spius Creek and Coldwater River. At relatively low spawner densities, the smaller sub-tributaries (Spius and Coldwater) produced higher fry densities for a given spawner density compared to the Nicola River mainstem, whereas at higher spawner densities, the opposite was true, at least for Spius Creek (Coldwater River lacked sufficiently higher spawner density data for this comparison).

ANCOVA with a model including spawner density and tributary (Table 10, model A) explained 68% of the variation in fry density, with 50% and 18% of the variation explained by spawner density effects ($P = 0.003$), and tributary effects ($P = 0.02$), respectively. This result is uncertain, however, owing to small sample sizes (4-7) for individual tributaries, and a lack of overlap in spawner density ranges for some tributaries (Figure 8). We could not discount the possibility that both fry and spawner densities depend on tributary; the data were too sparse to test for spatial autocorrelation (Dunham and Vinyard 1997). Spawner density \times tributary interactions were not significant ($P = 0.25$), although additional years of data and increased contrast in spawner abundance are needed to increase confidence in these inferences.

Since the data suggested that spawner abundance had a much greater effect on fry abundance than stream effects, we fit a Ricker stock-recruitment curve to pooled data for all tributaries (Figure 8, lower graph). The variance of the observations about this line represents process error among years in the stock-recruitment relationship for individual tributaries and variation in productivity and carrying capacity among tributaries. Observations for Deadman and Nicola Rivers and Spius Creek were negative outliers in relation to the Ricker curve for pooled tributaries (relatively low fry production per spawner), with the exception that relatively high spawner densities in Deadman and

Nicola Rivers produced positive outliers. Observations for Bonaparte and Coldwater rivers were, for the most part, positive outliers in relation to the Ricker curve for pooled tributaries. The stock-recruitment relationship for pooled data suggested that about 70 spawners per km produced a peak fry density of about 1,900 fry/km.

ANCOVA with a model including spawner density and year (Table 10, model B) explained 69% of the variation in fry density, and suggested strong spawner density effects ($P = 0.0003$), but no year effects ($P = 0.23$). It should be noted, however, that fry density data in 2004, which was lowest abundance year for all tributaries, was excluded from the model. Spawner density \times year interactions were not significant ($P = 0.14$). Results were similar for both models when adult and juvenile data were log-transformed prior to testing. Sample sizes were insufficient to test a model with all three effects (spawner density, tributary and year) included.

Although the data were very limited (3-6 years for individual tributaries), we found no evidence that brood fry density could be a reliable predictor of spawner recruitment the following generation. For this investigation, we assumed all fish from the brood year returned three years after the fry year, as data were unavailable to reconstruct adult returns by brood year for all the systems. About 90% of Chinook salmon in these systems mature and return to spawn three years following their initial year in freshwater (CTC 2002). Plots of spawner density versus brood fry abundance for individual streams did not suggest a positive association between the two (Figure 9). This analysis did not include the 2004 brood year when fry abundance was unusually low in all tributaries because the 2004 cohort will not return to spawn until 2007.

4.0 DISCUSSION

4.1 Efficiency of stratification and rapid assessment methods

The relatively precise overall estimates of Chinook fry abundance in the Thompson River basin from 2001 to 2006 (percent relative error of $\pm 16\%$ to $\pm 21\%$; Table 5, Appendix 3) can be attributed, in part, to the use of rapid assessment sampling methods that improved sampling efficiency. In tributaries reaches, sampling efficiency was also

improved by allocating higher sampling effort to run habitats that contained higher densities of fry than riffle habitats. This was somewhat fortuitous because the sampling design was intended to maximize sampling efficiency for steelhead parr, which were also more abundant in run habitats (Hagen et al. 2005). Higher sampling effort in runs resulted in higher precision, on average, for Chinook fry standing stock estimates for runs relative to riffles (runs: average = $\pm 47\%$, range = $\pm 13\%$ to $\pm 92\%$; riffles: average = $\pm 82\%$, range = $\pm 19\%$ to $\pm 320\%$; Appendix 3). The stratification of reaches in the Thompson River mainstem was also effective in addressing spatial variation in fry abundance because there were consistent differences in fry density among reach and habitat strata. The Thompson River mainstem accounted for about half of the total fry standing stock each year, and increasing sampling effort there relative to the tributaries would likely further improve the precision of the overall estimate.

High spatial variability in Chinook fry density in the study area necessitated sampling a relatively large number of sites to provide reasonably precise abundance estimates. Considering the size of the study area (462 km of habitat in seven streams), using multiple-pass depletion electrofishing would have been cost-prohibitive because of the time-consuming nature of this method. By using calibrated rapid assessment methods instead, we were able to increase sampling efficiency substantially. Field crews were typically able to conduct single-pass electrofishing at 2-3 sites in riffle habitats in an afternoon, snorkeling counts at an additional four sites in run habitats that evening. This compares to typical production of two sites per day for a two- or three-person crew employing conventional three-pass electrofishing.

The more important limitation of conventional electrofishing, however, is that it is unsuitable for assessing the abundance of juvenile salmonids in larger streams (Campbell and Neuner 1985; Hillman et al. 1992). If we had sampled only the smaller reaches in the our study area, and extrapolated observed fry densities observed there to the larger reaches, in most cases standing stocks in larger reaches would have been underestimated substantially (Table 5). In contrast, night snorkeler counts provided a versatile sampling method that could be applied to all habitats found in the lower Thompson River basin

despite considerable variation among reaches in channel width, gradient, substrate size and other characteristics. Because of its greater efficiency, the snorkeling survey methodology also allowed for habitats units in the larger streams, which could be up to 150 m long, to be sampled in their entirety, thereby reducing the sampling bias that may result from sampling only part of a heterogeneous habitat unit such as a pool (Hankin and Reeves 1988).

4.2 Calibration and accuracy of rapid assessment methods

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 entire watersheds including larger streams. The mark-recapture study indicated that, overall, night snorkel counts were reliable predictors of Chinook salmon fry abundance in the lower Thompson River basin. Snorkeling efficiency was consistently high (mean = 53%) and its error was normally distributed and relatively modest (95% CI = 48%-59%). Snorkeling efficiency was also largely unaffected by variation in physical habitat characteristics, underwater visibility, and stream temperature over the range in these variables represented by the mark-recapture sites. This is a highly desirable attribute for a sampling method because once adequate data is obtained to model the error distribution, the need to collect separate calibration data for each stream or habitat type is reduced. There is also the possibility of developing a regional model for snorkeling efficiency by combining calibration data from different studies. DFO currently uses a similar approach in conducting adult Chinook escapement surveys, whereby calibration data from key index streams are used to calibrate peak counts or AUC count data from data-limited systems (Parken et al. 2003, 2006). By process of elimination, the results of our mark-recapture study suggest variation in snorkeling efficiency was the result of variation in fish behaviour and observer efficiency among snorkelers.

The mark-recapture methodology used to calibrate snorkeling counts in this study relies on two critical yet untested assumptions. The first of these was that marked and

unmarked fish had equal probabilities of being seen by snorkelers, and the second was that marked fish did not leave the site during the 24-hour period between marking and the survey. Because these assumptions are difficult to evaluate directly, in the case of the first assumption, we relied on three-pass electrofishing as an independent calibration method for corroborating our mark-recapture estimates of snorkeling efficiency for run strata in the smaller tributaries. The higher average value for snorkeling efficiency estimated from three-pass electrofishing (70% versus 53%) suggests that, in the case of the first assumption, marked fish were not more visible than unmarked fish, barring the possibility that substantial numbers of them left the site. We could not detect marked fish that moved beyond the sections adjacent to the mark-recapture site that were also surveyed, but given that 96% of the 330 marked fish resighted by snorkelers had remained within the original marking section of the site, we found no evidence of large-scale movement from the marking sites. We could not discount the possibility that marked fish in the site modified their behaviour after being handled, and became more difficult to detect than unmarked fish 24 hours later during recapture swims. This could explain the lower snorkeling efficiency estimates for the mark-recapture calibration method versus the depletion electrofishing method. However, our fish handling protocol at mark-recapture sites was designed to minimize potential handling effects; fish were captured in a relatively low impact manner (hand nets), were not anaesthetized prior to marking, were released into their original holding location, and were allowed a 24-hour period of recovery prior to the snorkeling survey. Divers noted that after 24 hours marked fish occupied comparable locations to unmarked ones and behaved in a similar way. Qualitative observations of individual fish immediately after they were marked and released suggested that while marked fish did not necessarily remain in their original location, they selected holding locations comparable to those of unmarked fish, and made no obvious attempt to conceal themselves or flee downstream.

We could not eliminate the effects of open sites and fish behaviour as explanations for the discrepancy between mark-recapture and electrofishing-based snorkeling efficiency estimates. However, the alternative explanation, underestimation bias in depletion estimates, has been well documented. We assumed that depletion estimates represented

true fish abundance at the calibration sites. In fact, depletion estimates typically underestimate the actual number of fish present (Peterson and Cederholm 1984; Bohlin and Sundstrom 1977; Riley et al. 1993; Peterson et al. 2004), principally because the capture rate of fish remaining in the site declines with each subsequent pass. Negative bias in multiple-pass depletion estimates may be on the order of 15-25% or more even in ‘good’ estimates (see Bohlin and Cowx 1990). If our electrofishing snorkeling efficiency estimate is adjusted to account for 15-25% negative bias, the resulting adjusted estimate (53-60%) is in much better agreement with the mark-recapture-derived estimate of mean snorkeling efficiency (53%) for the same strata (runs in the smaller tributaries). During the concurrent steelhead parr snorkeling efficiency study, snorkeling efficiency estimates from the mark-recapture sites were also lower than snorkeling efficiency estimates derived from multiple-pass electrofishing, but were similar (72% for mark-recapture versus 67-73% for electrofishing) if the electrofishing-based estimates were adjusted for underestimation bias of 15-25% (Hagen et al. 2007 in prep.).

Single-pass electrofishing was a reliable predictor ($R^2 = 0.97$) of Chinook salmon fry abundance for shallow riffle strata in lower Thompson River tributaries. Although the number of calibration sites was limited (11), the proportion of fry captured on the first pass was high (mean = 78%) and relatively consistent (95% CI = 63-93%; Figure 2c), which helped to minimize error in the expansion factor applied to non-calibration sites. Other researchers have reported similar results for other species. For example, Jones and Stockwell (1995) used juvenile rainbow trout catches from a single electrofishing pass to successfully predict population estimates based on multiple depletions for sites in Ontario streams ($R^2 = 0.76-0.86$). Comparable relationships between single-pass electrofishing catches and multiple-pass depletion estimates have also been obtained for Atlantic salmon (Crozier and Kennedy 1994) and brown trout (Lobon-Cervia and Utrilla 1993). In our study, the potential for underestimation bias in three-pass depletion estimates applies to riffles as well as runs, which would lead to overestimation of capture efficiency and underestimation in fry standing stock in riffle strata that were sampled using single-pass electrofishing. However, we assumed that the degree of negative bias in the three-pass estimates would be relatively modest because habitat in the riffles appeared

conducive to effective electrofishing; riffle sites were generally shallow, lacked structure and complex cover (undercut banks and woody debris), were less than 20 m long, and were completely enclosed by block nets during sampling. As well, Chinook fry are not overly benthic in their orientation, and relatively swift water at riffle sites meant that stunned fry that were not netted were usually swept into the downstream block net. Regardless, error resulting from any negative bias in estimates of fry standing stock in riffles would have had little effect on the overall accuracy of fry population estimates because riffle strata sampled by electrofishing accounted for less than 20% and 4% of estimated yearly standing stocks for individual streams and the study area, respectively (Appendix 3).

The effectiveness of the rapid assessment methods employed in this study as suggested by the calibration relationships can be attributed to several factors. First was the excellent water clarity in the study area streams in October. In our experience, a minimum of 2 m horizontal visibility is necessary for divers to effectively count juvenile salmonids. Second, underwater counts were conducted by experienced teams of divers that used constant communication to avoid counting fish more than once. Third, stream habitat in the majority of the sampling strata (21 of 27 strata) in the study area was well suited to the method. The exceptions were shallow (< 30 cm mean water depth) riffles in the smaller tributaries. For these strata we substituted single-pass electrofishing, which is well suited to small streams and shallow habitats. In a study of snorkeling efficiency for bull trout and cutthroat trout (*O. clarki*) in small, shallow high gradient streams Thurow et al. (2006) reported a positive association with pool area, suggesting that adequate depth for divers to move systematically throughout the site may be an important determinant of snorkeling efficiency. Lastly, we conducted snorkeling surveys at night when juvenile salmonids occupied had moved to inshore locations (Bonneau and Scarnecchia 1998), and were typically resting on the substrate in a docile manner in areas of moderate to shallow depth and slow current. Other studies have also shown that underwater surveys are more effectively conducted at night (Thurow and Schill 1996; Bradford and Higgins 2000; Roni and Fayram 2000; Peterson et al. 2002).

4.3 Abundance, distribution, and size of fry

Multiple linear regression suggested that the most important predictor of fry abundance at the site level was distance upstream from the stream mouth, with sites further downstream having higher abundances. Chinook populations in other Fraser River tributaries (Murray and Rosenau 1989; Bradford and Taylor 1997) and in other river systems (Bjornn 1971), are known to disperse from tributary spawning areas to productive downstream rearing habitats. This may partly explain why fry densities were consistently higher at sites further downstream in individual streams. A similar, strong upstream-downstream gradient in fry density was also highly evident at the reach level: mean fry densities were higher in downstream versus upstream reaches in individual streams and for the lower Thompson River basin as a whole (Table 5). For example, the downstream reach of the Nicola River had fry densities that were 4-19 times higher than densities in the upstream reach, despite the fact that spawner densities in the downstream reach were about one tenth of those in the upstream reach (DFO SEDS database). In the Nicola River, the upper reach may offer high quality spawning habitat (Parken et al. 2003), but not fall-fry rearing habitat, whereas the opposite may be true for the lower reach.

Alternatively, higher fry densities in downstream reaches in the Thompson River basin in October may represent a protracted downstream migration through the system that begins shortly after emergence and culminates with smoltification and emigration to salt water the following spring. Based on patterns of Chinook fry abundance they observed in Kamloops Lake and the lower Thompson River mainstem, Beniston et al. (1985) posited that a portion of stream-type Chinook from North Thompson rear in Kamloops Lake during the summer and in the lower Thompson River during the fall. The considerably greater size (86 mm) of fry in the lowermost reach of the lower Thompson River compared to fry in the adjacent tributaries (64-79 mm) also suggests downstream movement given Bradford and Taylor's (1997) finding that larger Chinook fry are more likely to move downstream than smaller fry. Beniston et al. (1985) also attributed increasing fry size in downstream reaches in the lower Thompson River mainstem to downstream movement by larger individuals.

Habitat type, thalweg depth, proximate stream gradient, and stream were also significant predictors of fry density (Table 7). In the tributaries, fry densities were, on average, 10-fold higher in runs and pools than in riffles. In the Thompson River mainstem, fry densities were highest for the embayment habitat type, which, similar to runs and pools, represented relatively deep, low velocity habitats, relative to the other types that were delineated. The preference of Chinook for deeper habitats was also reflected in the positive association between fry density and thalweg depth at the site level. Other studies have also reported higher Chinook fry densities in relatively deep habitats (Everest and Chapman 1972; Swales et al. 1986; Ebersole et al. 2003). The negative association between fry density and proximate stream gradient could be explained by sampling sites located in lower gradient stream sections having a greater abundance of relatively deep, low velocity habitats compared to those in higher gradient sections.

Multiple linear regression also indicated that at sampling sites located in core habitats that supported relatively high fry densities, variation in abundance from one year to the next was lower compared to that for marginal habitats that supported relatively low fry densities. This suggests that as fry disperse from spawning sites and colonize downstream habitats, optimal habitats are selected first and are well occupied even in years of low overall abundance, whereas marginal sites may only support substantial fry densities in years of relatively high abundance.

The stock-recruitment analysis suggested that differences in fry densities among streams were driven primarily by differences in spawner densities. However, our results indicated that variation in productivity and productive capacity among streams influenced fry abundance as well. This is best illustrated by the broad range in predicted fry recruitment among streams for a given spawner density in Figure 8. Moreover, stream-type Chinook do not spawn in the Thompson River mainstem, yet we found that mean fry density across years in the Thompson was 2- to 5-fold higher than that in other nearby streams. Overall the Thompson mainstem supported 32% to 68% of the total standing

stock for the study area. This emphasizes the importance of including larger mainstem rearing habitats when designing sampling programs to estimate Chinook production within a river system, monitoring year-to-year trends in abundance, or planning habitat restoration programs. Our results indicated the lower stream reaches, often with low stream gradient, had the highest productive capacity for Chinook salmon fry.

Genetic samples collected in the middle and lower reaches of the Thompson River during October 2002 indicated that a substantial proportion (27% in 2002 and 52% in 2006) of the fall-fry standing stock were migrants from the adjacent tributaries. This corroborates the results of previous downstream trapping studies conducted in the Deadman River during July-September, 1978-79 (Starr 1979) and more recently during March-July 2001 (Mike Galesloot, Shuswap Nation Fisheries Commission, unpublished data), that recorded large numbers of Chinook fry emigrating from the Deadman River during spring and summer. We hesitate to make inferences based on limited genetic data collected from only two years. However, if one were to assume that our samples were representative of the proportional abundance of different Chinook stocks in the lower Thompson River in 2002 and 2006, one can speculate about the magnitude of fry migration from different tributaries to the Thompson mainstem. For example, if 10.9% and 34.4% (Table 4) of the estimated fry standing stock in the lower two reaches of the Thompson River in 2002 and 2006, respectively were from the Deadman and Bonaparte Rivers (56,200 and of 63,300 fry, respectively; derived from Table 5), then migrant fry would represent 15.5% and 20.3% of the Deadman/Bonaparte populations in 2002 and 2006, respectively, and this does not include the potential number migrants in the uppermost Thompson reach, directly downstream of the Deadman River confluence (Figure 1). A similar calculation for the Nicola basin suggests a comparable proportion of Thompson River immigrants comprising the Nicola standing stock (18.3% and 11.1% in 2002 and 2006, respectively). Furthermore, standing stock was not estimated for the Fraser River mainstem. We are uncertain of the proportion of the fry standing stock rearing there that originates from the study area.

Despite significant migration, however, the genetic data suggest that the majority of stream-type fry from lower Thompson River tributary populations rear in their natal tributaries. Surprisingly, despite their spawning areas being further away (80-200 km upstream), North Thompson Chinook were similarly or more abundant in the lower Thompson River mainstem than fry from lower Thompson tributaries (genetic samples from the lower Thompson River mainstem were comprised of 73% and 27% North Thompson fry in 2002 and 2006, respectively). North Thompson Chinook would appear more likely to disperse to downstream rearing habitats than Chinook from lower Thompson tributaries. Consider that for 2002 fry standing stocks, the total brood escapement was 8,588 for the three populations (North Thompson mainstem, Clearwater and Barrier; DFO SEDS database) that represented the bulk (93%) of North Thompson spawning stock. This was less than half that for the lower Thompson tributaries (17,521 spawners), yet, our genetic data suggested that North Thompson fry accounted for the majority of the fry standing stock in the lower Thompson mainstem. Beniston (1985) observed a 94% decline between June-August in the relative abundance of Chinook fry in Kamloops Lake (headwaters of the lower Thompson River, Figure 1), which he assumed were from the North Thompson River⁴, and he attributed this decline to subsequent movement to the lower Thompson mainstem, a contention that was supported by his observations of the seasonal patterns in mean fry length in the lower Thompson mainstem. Lower quality rearing conditions in the North Thompson River, particularly in the mainstem which is subject to high glacial flows during much of the growing season (Parken et al. 2002), may explain why North Thompson fry are more likely to migrate to the lower Thompson mainstem. By comparison, higher water temperatures and lower turbidity in the lower Thompson tributaries provide a better rearing environment.

Currently, Chinook populations in the lower Thompson and North Thompson rivers are considered to have equivalent 'stream-type' early life histories, spending one year (rarely two years) in freshwater prior to ocean migration (Candy et al. 2002). However,

⁴ Beniston (1985) based his assumption on the fact that most South Thompson River populations were ocean-type, or that South Thompson fry were much larger in size (mean = 75 mm) by June than both North Thompson River fry (50 mm) and fry captured in Kamloops Lake (52 mm).

as discussed above these two groups appear to differ considerably with regard to their propensity to disperse to non-natal downstream rearing habitats. This may be important to consider in determining escapement goals for each group. With respect to the mainstem spawning population in the lower Thompson River and the numerous populations in the South Thompson River, the genetic sampling results from this study support the findings of Candy et al. (2002) that these stocks are primarily ‘ocean-type’ (rearing in freshwater for 60-150 days prior to ocean migration) as none of the fry samples collected in the lower Thompson River in October were identified as belonging to these populations.

Our comparisons among years of standing stocks of steelhead parr and fry with Chinook salmon fry standing stocks provide little evidence that Chinook abundance is a factor limiting steelhead production, or vice versa. In fact, a significant and relatively strong positive correlation in shallow riffle habitats ($r = 0.85$, $P = 0.034$) between Chinook fry and steelhead fry standing stocks suggests that in the lower Thompson basin similar density-independent environmental factors affect survival in both species. Research by Everest and Chapman (1972) suggested that juvenile Chinook and steelhead of the same size used the same physical space in streams, but very different emergence timing resulted in body size differences that limited the potential for interspecific competition. Importantly, in their study steelhead did not change their habitat preference in the presence of Chinook (interactive segregation) a typical indication of the presence of interspecific competition.

4.4 Relationship between fry abundance and spawner abundance

Our stock-recruitment analyses for individual tributaries were limited by few years of data (5-9 years) and low contrast in spawner abundances, and may not accurately reflect true relationships between Chinook adult and fry abundance. Nevertheless, in most cases, the range in escapement for each tributary was relatively broad, and within that range, fry abundance was positively related to brood spawner abundance. Differences in productivity among tributaries were evident, but an analysis of pooled data from all tributaries indicated that spawner abundance still explained the majority of the variation

in fry abundance (Table 10). Spius Creek and Coldwater River showed evidence of density-dependent fry survival at higher escapement levels in recent years, suggesting that S_{msy} may have been met or exceeded, whereas Nicola and Deadman rivers exhibited a linear relationship between adult and fry abundance suggesting that increased escapements in recent years still have not achieved S_{msy} . The fact that the size of fall fry in the tributaries was negatively associated with fry abundance (Figure 4) also suggests that density dependent ecological processes were occurring at escapement levels observed during the study.

The benefits of acquiring reliable stock-recruitment data for tributary stream-type Chinook stocks in the lower Thompson River basin are substantial. For one, adult-to-fry stock-recruitments can be used to evaluate predictions of S_{msy} for Interior Fraser stream-type Chinook stocks based on an existing watershed area model (Parken et al. 2006). This is of particular importance considering that Interior Fraser streams are not represented in the dataset used to develop the model. Our adult-to-fry data suggest that the watershed area model substantially overestimated S_{msy} for Spius Creek (2000 spawners; 80% confidence interval: 1,600-2,400) versus ≈ 600 -800 spawners for our adult-to-fry stock-recruitment analysis (Figure 7), which was consistent with interpretations of Parken et al. (2006). Moreover, our analysis included adults from the Nicola stock that spawn in the lower reach of Spius Creek because of difficulties discerning fry production from the early and late stock components, whereas Parken et al. (2006) did not include lower Spius Creek in their estimate of watershed area. Spius Creek is relatively high in gradient and may have less available spawning habitat for its watershed area compared to the other tributaries in our study (see Figure 14 in Parken et al. 2002). For Coldwater River our data suggests an S_{msy} of ≈ 600 -900 spawners for the entire stream including Nicola spawners in the lower reach compared to the watershed area model S_{msy} prediction of 1,100 adults (80% confidence interval: 900-1,400) for the early Spring stock in the upper reach alone.

Our Ricker stock-recruitment models for Spius Creek and Coldwater River stocks had higher estimates of recruits-per-spawner (a parameter) at low stock sizes (Table 9) than

for other lower Thompson River tributaries, but this result does not have a clear interpretation. It is possible that these streams provide relatively high quality spawning and rearing habitat, but only in localized areas, whereas the larger streams (Bonaparte and Nicola Rivers) have suitable spawning and rearing habitat throughout, although not of necessarily the same quality. Thus, when spawner and fry densities are expressed as averages across the entire stream length, the larger streams have a higher carrying capacity for both. Our data showed little to weak evidence of density dependent fry survival in Deadman and Nicola Rivers at escapement levels below S_{msy} values predicted by the watershed area model of Parken et al. (2006). However, if the true S_{msy} was near the habitat model prediction or higher, then a positive linear relationship would be expected at these escapement levels. Fry surveys in years of higher escapement would be beneficial for determining fry carrying capacity in these relatively productive streams. With respect to the relatively high estimates of the a parameter for Spius Creek and Coldwater River, Hilborn and Walters (1992) noted that large errors of estimation for the spawning stock size has the effect of elongating the data cluster, thereby resulting in an overestimate of the slope of the recruitment curve at low spawning stock sizes.

We recognize several limitations to our adult-to-fry stock-recruitment analysis. Firstly, we were unable to partition fry numbers among early and late run stocks that spawn in the upper and lower reaches of Coldwater River and Spius Creek, respectively. Separate escapement estimates were available for these stocks, but given the propensity of juvenile Chinook to migrate downstream, the fry population in the lower reaches of these streams in October is likely a mix of the two. In estimating stock-recruitment parameters for these streams, we combined adult and fry numbers from the early- and late-run Chinook, but this may have biased our results if productivity differs for the two stocks. Secondly, we omitted reaches that likely contribute to fry production for lower Thompson River stocks (e.g., Nicola River upstream of Coldwater River, Maka Creek), resulting in negative bias in our estimates of fry carrying capacity and fry numbers per spawners for some stocks. Lastly, we were unable to assess non-natal fry production for individual stocks, which also resulted in negative bias in our estimates of fry carrying capacity and productivity. Genetic sampling indicated that stream-type Chinook fry in

lower Thompson River tributaries also rear extensively in the Thompson River mainstem. They may rear in the Fraser River and its tributaries as well (Murray and Rosenau 1989). We were unable to estimate the number of fry from individual tributaries that were present in the Thompson River mainstem each year because genetic data was only collected in 2002 and 2006. These limitations may partly explain why our adult-to-fry stock-recruitment analyses suggested lower S_{msy} values compared to the watershed area model of Parken et al. (2006). Their model was developed using adult-adult stock-recruitment data, and therefore integrated fry production from non-natal habitats. Measurement error in either escapement or fry abundance is another potential source of bias in our analysis. The relatively steep recruitment curves for Spius Creek and Coldwater River, compared to other streams (Figure 8), were influenced by low abundance data collected as part of earlier work in the 1980's. Methods used for estimating fry standing stock and escapement at that time were considerably less rigorous than our approaches from 2001-2006 (Table 7).

As adult-to-fry stock-recruitment data accumulate for lower Thompson River tributaries, it may be possible to further refine the watershed area model for these and other interior Fraser streams. Discriminating amongst stock-recruitment models (Ricker, Beverton-Holt, etc.) can also give important insight into the freshwater production dynamics of Chinook salmon, and mechanisms of density dependent survival. Freshwater production of coho salmon and steelhead smolts appears to be strongly asymptotic, and stock-recruitment dynamics for these species are generally better described by the Beverton-Holt relationship (Bradford et al. 2000; Ward and Slaney 1993). In contrast, the relatively moderate slopes of the Chinook adult-to-fry stock-recruitment relationships suggested by our data, in particular for Deadman and Nicola Rivers, supports previous contentions that the Ricker relationship is more appropriate for modeling stock-recruitment dynamics for Chinook salmon (CTC 1999).

Our stock-recruitment analysis differed from that of Parken et al. (2006) and the above-mentioned studies for steelhead and coho in that we used young-of-the-year fry abundance as the measure of recruitment as opposed to adult or smolt abundance,

respectively. When fry abundance is substituted for adult abundance as the index of recruitment, the confounding effects of variation in marine survival and exploitation are eliminated. However, adult-to-fry stock-recruitment analyses will likely produce higher estimates of S_{msy} and S_{rep} relative to adult-to-adult or adult-to-smolt stock-recruitment analyses if significant density dependent bottlenecks to Chinook production exist at stages in the life cycle beyond the fall fry stage (e.g., overwinter survival in freshwater, Ward and Slaney 1993). For steelhead and Atlantic salmon, escapements that maximize smolt production are usually lower than escapements that maximize fry production (Ward and Slaney 1993; Symons 1979). The fact that our analysis did not consistently suggest higher estimates of S_{msy} than those predicted by the Parken et al. (2006) watershed area model points to the possibility that freshwater production of stream-type Chinook in the lower Thompson River is constrained more by spawning habitat or egg-to-fry survival rather than fry-to-smolt survival.

The cause of unusually low egg-to-fry survival for the 2004 fry cohort in all study tributaries was unclear. One possibility was unfavorable spawning conditions. Of the brood years, spawners in 2003 (2004 brood year) were subject to the most severe low streamflow/high temperature conditions during their final migration and spawning period (WSC streamflow data on file, MOE and DFO water temperature data on file, and this may have led to unusually poor spawning success and egg survival (Quinn et al. 2007). While this remains uncertain, unfavorable environmental conditions in the lower Thompson River can result in very high, density independent egg-to-fry mortality regardless of escapement abundance. This has important implications with regard to forecasting adult returns, stock status, and fishery management. Although our very limited data did not suggest a strong association between brood fry abundance and subsequent adult escapement (Figure 9), very low fry abundance in all the tributaries in 2004 suggests the possibility of low adult returns in 2007, despite relatively high brood escapement. Monitoring fry abundance annually in conjunction with existing escapement monitoring programs could allow managers to respond to unusually low or high egg-to-fry survival by predicting adult returns.

The quantification of measurement error in production estimates is necessary for stock-recruitment analysis. The results of this study suggest that measurement error in fry abundance can be reliably estimated and therefore corrected for. Quantifying measurement error is necessary for two reasons. First, unequal variances among estimates of production at different stock sizes violate the assumption of homoscedasticity in linear regression. In practice violation of the equal variances assumption is common only when two or more methods of estimating production are employed (CTC 1999), but this was the case with our stock-recruitment data set because of the incorporation of fry standing stock data from other, earlier studies (Table 8). Although beyond the scope of our analysis, weighted regression can address inequality in relative sampling error across years (CTC 1999). Measurement error also affects estimates of S_{msy} by confounding estimates of the error associated with process error. Estimates of S_{msy} should be corrected for the presence of process error, but not for measurement error (CTC 1999).

A critical requirement for reliable stock-recruitment analysis is that enough data exists for parameter estimates from the analysis to be unbiased and reasonably precise. Our largest dataset was nine years of adult and fry abundance estimates for Coldwater River (Table 8), which was still well short of the minimum 15 years recommended by the Joint Chinook Technical Committee (CTC 1999). In addition, stock-recruitment analysis requires that the range of stock-recruitment data include stock sizes over which density dependent mortality factors occur. Brood year spawner abundances were likely not high enough to reveal density-dependent recruitment in Nicola and Deadman rivers, and were not high enough in all tributaries to estimate S_{rep} or to determine which stock-recruitment model (Ricker or Beverton-Holt) best described the stock-recruitment relationship.

A final requirement is sufficient contrast in brood year spawner escapements for the stock-recruitment relationship to be discerned. The Joint Chinook Technical Committee of the Pacific Salmon condition recommends a contrast between highest and lowest escapements of eight times (CTC 1999), which occurred for Spius Creek and Deadman and Coldwater rivers in our study (Table 8). However, for the Nicola, Nicola aggregate,

and Bonaparte stocks, contrast between highest and lowest escapements was only 6.0, 6.4, and 2.4 times, respectively.

Our analysis of available adult and fry data for stream-type Chinook in lower Thompson River tributaries indicates the potential for informative stock-recruitment analyses. Collecting additional fry data should therefore be pursued, given that escapement data will continue to be collected by DFO monitoring programs. Consideration should also be given to managing for higher escapements to these streams relative to escapements observed during this study, so that the adult-to-fry stock-recruitment relationships can be estimated and to better evaluate the utility of habitat-based approaches to generating escapement goals.

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Table 1. Watershed area, flow characteristics, wetted widths and accessible stream length for Chinook salmon for the lower Thompson River and its major tributaries.

Stream	Watershed area (km ²)	Stream length incl. in study area (km)	Mean annual discharge (m ³ /s) ¹	Mean October discharge (m ³ /s)	Mean October wetted width (m)
Bonaparte	5020	108	5.7 ²	3.3 ²	12.1
Coldwater	914	79	8.2	2.6	12.1
Deadman	1450	36	3.5	1.1	10.4
Nicola	7280	75 ³	27.1	8.2	21.1
Spius	780	39	10.4	2.7	12.2
Thompson	-	125	743.4	470.0	98.9

¹ Water Survey of Canada data on file

² In years after the construction of a discharge control structure at Bonaparte Lake in 1994

³ Only the portion of Nicola River from the Coldwater River confluence to the Thompson River confluence was included in the study; Chinook also spawn in the upper Nicola River above Nicola Lake, and in the reach between Nicola Lake and Coldwater River.

Table 2. Description of habitat stratification and juvenile Chinook sampling in the lower Thompson River basin during 2001-2006. For each stratum, the sampling and calibrated method are given. Also shown are total strata lengths, average site lengths, and the number of sites sampled in each strata in each year.

Stream reach	Habitat type	Total length (km)	Sample method	Calibration method	Mean site length (m)	Sample size					
						2001	2002	2003	2004	2005	2006
L. Bonaparte (B1)	deep riffle	15.8	snorkeling	mark recap.	21	5	6	6	6	NS ¹	6
	run	43.0	snorkeling	mark recap.	43	10	12	12	12	NS ¹	12
U. Bonaparte (B2)	deep riffle	24.5	snorkeling	mark recap.	17	2	2	2	2	NS ¹	2
	run	24.5	snorkeling	mark recap.	38	4	4	4	4	NS ¹	4
L. Coldwater (CW1)	shallow riffle	15.6	1-pass EF	3-pass EF	14	3	3	3	3	NS ¹	2
	run	12.3	snorkeling	mark recap.	44	5	5	5	6	6	5
U. Coldwater (CW2)	shallow riffle	28.5	1-pass EF	3-pass EF	16	5	6	5	5	NS ¹	5
	run	22.6	snorkeling	mark recap.	32	11	11	11	9	10	10
Deadman (D)	shallow riffle	16.8	1-pass EF	3-pass EF	14	5	5	6	6	6	5
	run	19.3	snorkeling	mark recap.	39	10	11	11	11	12	11
L. Nicola (N1)	deep riffle	27.8	snorkeling	mark recap.	20	8	9	8	9	9	9
	run	22.8	snorkeling	mark recap.	75	11	10	9	9	10	10
U. Nicola (N2)	shallow riffle	10.2	1-pass EF	3-pass EF	18	3	3	4	3	4	3
	run	13.8	snorkeling	mark recap.	63	5	4	5	6	5	5
L. Spius (SP1)	shallow riffle	4.4	1-pass EF	3-pass EF	12	3	6	5	6	NS ¹	6
	run	2.3	snorkeling	mark recap.	35	4	6	6	6	6	6
U. Spius (SP2)	shallow riffle	19.3	1-pass EF	3-pass EF	15	4	4	4	5	4	4
	run	12.9	snorkeling	mark recap.	16	4	5	4	5	5	6
Thomp. (T1)	bar	8.3	snorkeling	mark recap.	81	4	3	3	3	3	3
	dfs	11.4	snorkeling	mark recap.	82	4	3	3	3	3	3
	embay	19.8	snorkeling	mark recap.	69	4	3	3	3	3	3
	rapid	0.2	not sampled	-	-	-	-	-	-	-	-
Thomp. (T2)	bar	14.4	snorkeling	mark recap.	79	6	4	4	4	4	4
	dfs	19.9	snorkeling	mark recap.	85	6	4	4	4	4	4
	embay	19.8	snorkeling	mark recap.	73	5	4	4	4	4	4
	rapid	2.8	not sampled	-	-	-	-	-	-	-	-
Thomp. (T3)	bar	3.6	snorkeling	mark recap.	88	4	4	4	4	4	4
	dfs	10.6	snorkeling	mark recap.	85	4	4	4	4	4	4
	embay	11.9	snorkeling	mark recap.	81	4	4	4	4	4	4
	rapid	2.1	not sampled	-	-	-	-	-	-	-	-
Total		461.2				143	145	143	146	110	144

¹ Not sampled due to poor survey conditions

Table 3. Result for calibration models for expanding snorkeling counts and single-pass electrofishing catches to provide estimates of Chinook fry abundance at individual sampling sites in the lower Thompson River study area. Mark-recapture was used to calibrate snorkeling counts for all strata sampling using this method. Three-pass electrofishing depletion estimates were used to calibrate single-pass electrofishing catches for shallow riffle strata in small tributaries. Model 3 was not used to calibrate snorkeling counts of Chinook fry in this study, and was developed only to provide a second, independent model of snorkeling efficiency for comparison with model 1.

Model	Sampling method	Sampling strata ¹	Calibration method	Mean sampling efficiency	<i>n</i>	95% CI
1	Snorkeling	All strata sampled by snorkeling	MR	53%	40	48%-59%
2	1-pass EF	Shallow riffles (small tribs)	3-pass EF	78%	11	63%-93%
3	Snorkeling	Runs (small tribs)	3-pass EF	70%	18 ²	47%-92%

¹ See Table 2

² After removal of two outliers (see Section 3.3)

Table 4. Proportional (%) representation of parent Chinook stocks from the lower Thompson and North Thompson River basins in Chinook fry samples collected in the middle (T2) and lower (T3) reaches of the lower Thompson River mainstem in October 2002 and 2006. Standard errors of the estimates are given in brackets.

Parent stock	2002			2006		
	Reach T2 <i>n</i> = 47	Reach T3 <i>n</i> = 50	Pooled data ¹ <i>n</i> = 97	Reach T2 <i>n</i> = 62	Reach T3 <i>n</i> = 66	Pooled data ² <i>n</i> = 128
<u>Lower Thompson system</u>						
Deadman and Bonaparte	13.6 (5.9)	5.5 (5.2)	10.9	57.3 (8.8)	13.0 (5.3)	34.4
L. Thompson mainstem	0.0 (0.3)	0.0 (1.6)	0.0	0.0 (0.4)	0.0 (0.2)	0.0
Nicola system ²	9.5 -	26.6 -	16.7	0.5 -	32.8 -	17.2
- Nicola	0.0 (1.6)	13.4 (9.5)	3.3	0.1 (0.9)	27.6 (7.1)	14.3
- Upper Coldwater	9.0 (5.6)	12.8 (5.6)	12.6	0.1 (0.7)	0.7 (2.4)	0.4
- Upper Spius	0.5 (2.3)	0.4 (3.4)	0.8	0.3 (1.0)	4.4 (5.4)	2.4
<u>North Thompson system</u>						
Barriere	13.3 (8.2)	12.1 (7.3)	13.3	16.6 (4.9)	2.7 (4.4)	9.4
Blue	5.1 (1.5)	0.1 (1.3)	2.1	0.0 (0.3)	0.0 (0.3)	0.0
Clearwater	10.3 (6.0)	6.6 (7.3)	9.2	9.4 (6.0)	12.2 (4.8)	10.9
Lemieux	0.0 (1.7)	1.8 (2.7)	0.0	0.1 0.461	0.2 (0.9)	0.1
Mahood	0.0 (2.2)	2.7 (3.1)	0.5	0.0 (0.3)	0.0 (0.3)	0.0
N. Thompson mainstem	43.4 (12.8)	44.4 (11.0)	45.1	13.0 (6.3)	34.8 (7.6)	24.2
Raft	3.8 (9.6)	0.0 (4.9)	2.1	0.2 (1.1)	1.8 (3.3)	1.0
<u>Totals</u>						
Lower Thompson tribs	23.0 (6.5)	32.1 (7.6)	27.2	58.0 (8.6)	45.9 (6.1)	51.7
North Thompson system	75.8 (8.6)	67.7 (9.8)	72.7	39.4 (8.4)	51.8 (6.3)	45.8
South Thompson system	0.0 (3.2)	0.0 (2.8)	0.0	0.5 (1.1)	0.3 (0.8)	0.4
Misidentified ³	1.2 -	0.2 -	0.1	2.1 -	2.0 -	2.1

¹ Maximum likelihood model was run using combined data from the two reaches.

² Maximum likelihood model was not run for the combined data; **pooled estimates were computed as the average for the two reaches, weighted by sample size.**

³ Likelihood model categorized several samples as being from parent stocks from streams far removed from the lower Thompson River (lower and middle Fraser River stocks)

Table 5. Estimated Chinook fry standing stocks by reach and stream, and for the study area during 2001-2006. Linear fry densities and percent relative errors (average 95% confidence interval divided by the mean) for standing stock estimates are shown also.

Stream or reach	Standing Stock						Density (fry/km)							Percent relative error (95% CI/estimate)					
	2001	2002	2003	2004	2005	2006	2001	2002	2003	2004	2005	2006	mean	2001	2002	2003	2004	2005	2006
Lower Bonap.	182,953	206,201	169,860	23,827	ND ¹	191,917	3,111	3,507	2,889	405	n/a	3,264	2,635	37%	32%	49%	50%	n/a	38%
Upper Bonap.	20,857	66,257	67,682	26,510	ND	34,548	426	1,352	1,381	541	n/a	705	881	63%	50%	54%	45%	n/a	58%
Bonaparte	203,810	272,457	237,541	50,336	ND	226,465	1,891	2,527	2,204	467	n/a	2,101	1,838	33%	27%	39%	35%	n/a	34%
Lower Cold.	45,779	82,769	40,430	22,403	70125 ²	51,310	1,648	2,837	1,495	787	2,324	1,821	1,819	44%	47%	81%	32%	48%	35%
Upper Cold.	57,786	43,820	41,718	15,589	22648 ²	45,077	946	887	825	408	459	781	718	47%	22%	43%	73%	46%	43%
Coldwater	103,566	126,588	82,148	37,992	92774 ²	96,387	1,194	1,576	1,062	542	1,117	1,148	1,106	32%	32%	47%	36%	35%	27%
Deadman	15,463	34,320	78,450	8,736	56,945	21,618	428	949	2,170	242	1,575	598	994	57%	103%	72%	74%	64%	79%
Lower Nicola	85,766	204,651	173,928	38,667	109,088	107,884	1,695	4,044	3,437	764	2,156	2,132	2,371	33%	29%	33%	27%	30%	38%
Upper Nicola	16,888	28,793	41,625	4,590	5,653	18,150	704	1,200	1,734	191	236	756	803	55%	44%	47%	44%	53%	47%
Nicola	102,654	233,444	215,553	43,257	114,741	126,035	1,376	3,129	2,889	580	1,538	1,689	1,867	29%	26%	27%	24%	29%	33%
Lower Spius	16,325	15,841	18,607	5,926	10675 ²	17,986	2,437	2,364	2,777	884	1,593	2,684	2,123	45%	68%	30%	36%	41%	28%
Upper Spius	40,977	9,150	22,201	1,411	16,604	13,680	1,273	284	689	44	516	425	538	46%	84%	75%	134%	80%	76%
Spius	57,302	24,991	40,808	7,337	27278 ²	31,666	1,473	642	1,049	189	701	814	811	36%	54%	44%	43%	42%	36%
Thomp. (T1)	53,379	81,214	68,547	61,375	34,856	47,491	1,350	2,054	1,734	1,552	882	1,201	1,462	31%	45%	58%	64%	42%	52%
Thomp. (T2)	115,564	321,177	517,673	186,609	182,452	135,011	2,136	5,936	9,567	3,449	3,372	2,495	4,492	45%	49%	55%	28%	27%	46%
Thomp. (T3)	193,124	194,314	226,009	75,580	136,353	49,093	7,404	7,449	8,664	2,897	5,227	1,882	5,587	71%	47%	34%	27%	36%	42%
Thompson	362,067	596,705	812,229	323,564	353,661	231,596	3,024	4,984	6,784	2,702	2,954	1,934	3,730	41%	32%	37%	21%	20%	30%
Total or mean	835,609	1,286,388	1,468,472	476,042	770,850 ²	728,093	1,832	2,820	3,219	1,044	1,690	1,596	2,033	21%	17%	22%	15%	18%	16%

¹ ND indicates no data were collected for a particular reach/stream due to high flows

² Chinook fry standing stocks were approximated for missing strata to compute these values (see Section 3.6)

Table 6. Pearson correlation matrix (*r* values) for annual Chinook fry standing stocks in streams of the lower Thompson River basin during 2001-2006.

	Bonaparte	Coldwater	Deadman	Nicola	Spius
Bonaparte	1.000				
Coldwater	0.912	1.000			
Deadman	0.527	0.175	1.000		
Nicola	0.861	0.695	0.764	1.000	
Spius	0.564	0.533	0.268	0.246	1.000
Thompson	0.456	0.211	0.909	0.790	0.177

Table 7. Standard multiple linear regression of continuous and categorical physical habitat variables on mean density (fish/km) of Chinook fry among sites in the lower Thompson River basin during 2001-2006. See section 3.7 for a description of the independent variables.

Variable	Correlation with fry				
	density	<i>B</i>	<i>sr</i> ² (unique)	<i>F</i>	<i>P</i>
lnD90	0.091	0.243	0.004	1.685	0.196
lnFINESUB	0.169	0.058	0.001	0.493	0.483
lnDEPTH	0.436	0.718	0.012	5.587	0.019
lnTURB	-0.132	0.03	<0.001	0.162	0.688
lnLWD	-0.019	-0.028	<0.001	0.05	0.823
STREAM	-	-	0.009	4.044	0.002
HABTYPE	-	-	0.039	18.327	<0.001
DISTANCE	-0.197	1.052	0.118	56.019	<0.001
GRADIENT	-0.319	-0.025	0.050	23.918	<0.001

$R^2 = 0.59$
 $R = 0.765^{**}$

** $P < 0.001$

Table 8. Summary of Chinook brood escapements, and resultant fry recruitment and spawner recruitment for tributaries of the lower Thompson River for years included in this study (2001-2006), and previous years for which data were available.

Stream	Fry year	Brood escapement ¹	Fry recruitment	Spawner recruitment ²	Source for previous fry data	Method of estimation for spawner numbers
Bonaparte	2001	5,347	203,810	8,035	n/a	direct count (fence)
Bonaparte	2002	6,285	272,457	7,990	n/a	direct count (fence)
Bonaparte	2003	8,368	237,541	3,527	n/a	direct count (fence)
Bonaparte	2004	8,035	50,336	n/a	n/a	direct count (fence)
Bonaparte	2005	7,990	ND ³	n/a	n/a	direct count (fence)
Bonaparte	2006	3,527	226,465	n/a	n/a	direct count (fence)
Coldwater	1978	600	68,463	200	Wightman 1979	peak count
Coldwater	1980	300	40,000	547	Sebastion 1982	peak count
Coldwater	1981	710	5,000	598	Sebastion 1982	peak count
Coldwater	2001	1,362	103,566	1,687	n/a	mark-recapture and peak count
Coldwater	2002	1,567	126,588	1,519	n/a	mark-recapture and peak count
Coldwater	2003	2,126	82,148	277	n/a	mark-recapture and peak count
Coldwater	2004	1,687	37,992	n/a	n/a	mark-recapture and peak count
Coldwater	2005	1,519	92,774	n/a	n/a	mark-recapture and peak count
Coldwater	2006	277	96,387	n/a	n/a	mark-recapture and peak count
Deadman	1979	280	11,400	600	Tredger 1980b	peak count
Deadman	1991	210	15,872	1,591	Olmstead et al. 1992	direct count (fence)
Deadman	2001	770	15,463	1,079	n/a	direct count (resistivity counter)
Deadman	2002	770	34,320	1,159	n/a	direct count (resistivity counter)
Deadman	2003	1,890	78,450	417	n/a	direct count (resistivity counter)
Deadman	2004	1,050	8,736	n/a	n/a	direct count (resistivity counter)
Deadman	2005	1,120	56,945	n/a	n/a	direct count (resistivity counter)
Deadman	2006	420	21,618	n/a	n/a	direct count (resistivity counter)
Nicola	1980	2,300	73,000	1,800	Sebastion 1982	peak count
Nicola	1981	5,000	64,200	3,700	Sebastion 1982	peak count
Nicola	2001	6,788	102,654	13,843	n/a	mark-recapture
Nicola	2002	8,182	233,444	9,254	n/a	mark-recapture
Nicola	2003	11,494	215,553	3,013	n/a	mark-recapture
Nicola	2004	13,843	43,257	n/a	n/a	mark-recapture
Nicola	2005	9,254	114,741	n/a	n/a	mark-recapture
Nicola	2006	3,013	126,035	n/a	n/a	mark-recapture
Spius	1980	50	13,250	102	Sebastion 1982	peak count
Spius	1981	200	5,400	256	Sebastion 1982	peak count
Spius	2001	1,123	57,302	1,412	n/a	mark-recapture and peak count
Spius	2002	842	24,991	2,373	n/a	mark-recapture and peak count
Spius	2003	1,476	40,808	393	n/a	mark-recapture and peak count
Spius	2004	1,412	7,337	n/a	n/a	mark-recapture and peak count
Spius	2005	2,373	27,279	n/a	n/a	mark-recapture and peak count
Spius	2006	393	31,666	n/a	n/a	mark-recapture and peak count

¹ Subtract one year for actual escapement year

² Add three years for actual escapement year

³ No data were collected for Bonaparte River in 2005 due to high flows

Table 9. Summary of parameter estimates for Ricker and Beverton-Holt models of Chinook adult-to-fry stock-recruitment relationships in tributaries of the lower Thompson River basin.

Stream	Spawner densities (per km)	Ricker parameters		Beverton-Holt parameters	
		<i>a</i>	<i>b</i>	<i>a</i>	<i>b</i>
Bonaparte ¹	33-78	-	0.024	2034	-
Deadman	6-54	48	0.006	9002	191.0
Coldwater	4-27	252	0.069	1401	2.9
Spius	1-61	100	0.036	962	5.3
Nicola	31-186	34	0.006	3236	73.0
Nicola total	14-88	50	0.012	2142	33.0
All tributaries	6-186	75	0.014	1868	13.8

¹ No stock-recruitment data available at low spawner densities

Table 10. Analysis of covariance tables for models of the relationships between fry density and (A) spawner density with qualitative tributary effects, and (B) between fry density and spawner density with qualitative year effects.

Source	df	MS	<i>F</i>	<i>R</i> ²	<i>P</i>
A: Spawner density and tributary effects				0.68	
Spawner density	1	4010120.0	17.94		0.002
Tributary	4	795256.1	3.56		0.053
Error	26	223473.8			
B: Spawner density and year effects				0.74	
Spawner density	1	5090520.0	19.25		0.0003
Year	9	381971.9	1.44		0.232
Error	12	264457.0			

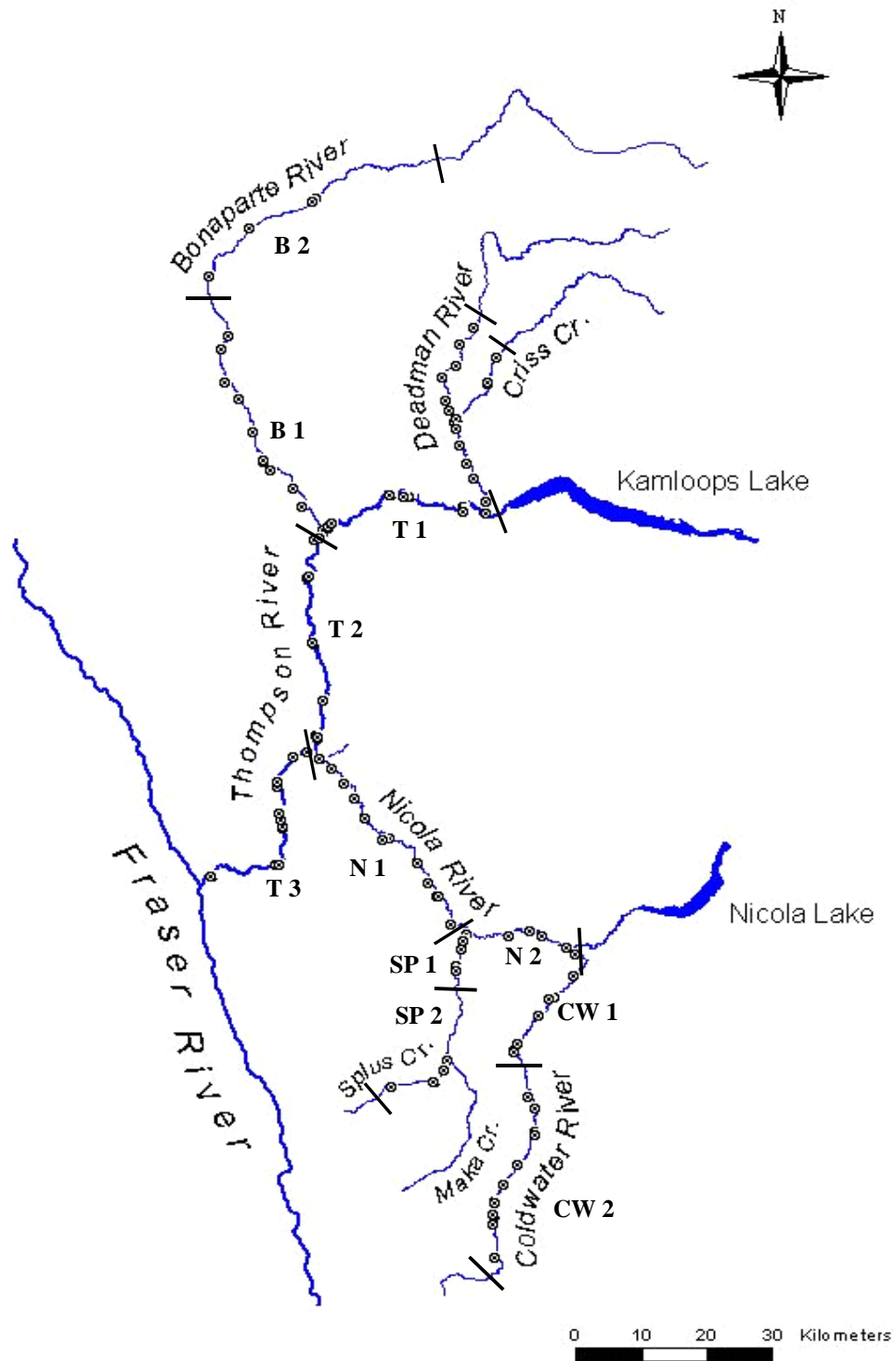


Figure 1. Map of the lower Thompson River basin showing all mainstem and tributary reaches included in the study area. Reach names are given for streams with more than one reach. Reach breaks are indicated by solid slashes and Chinook fry sampling sites are indicated by dotted circles.

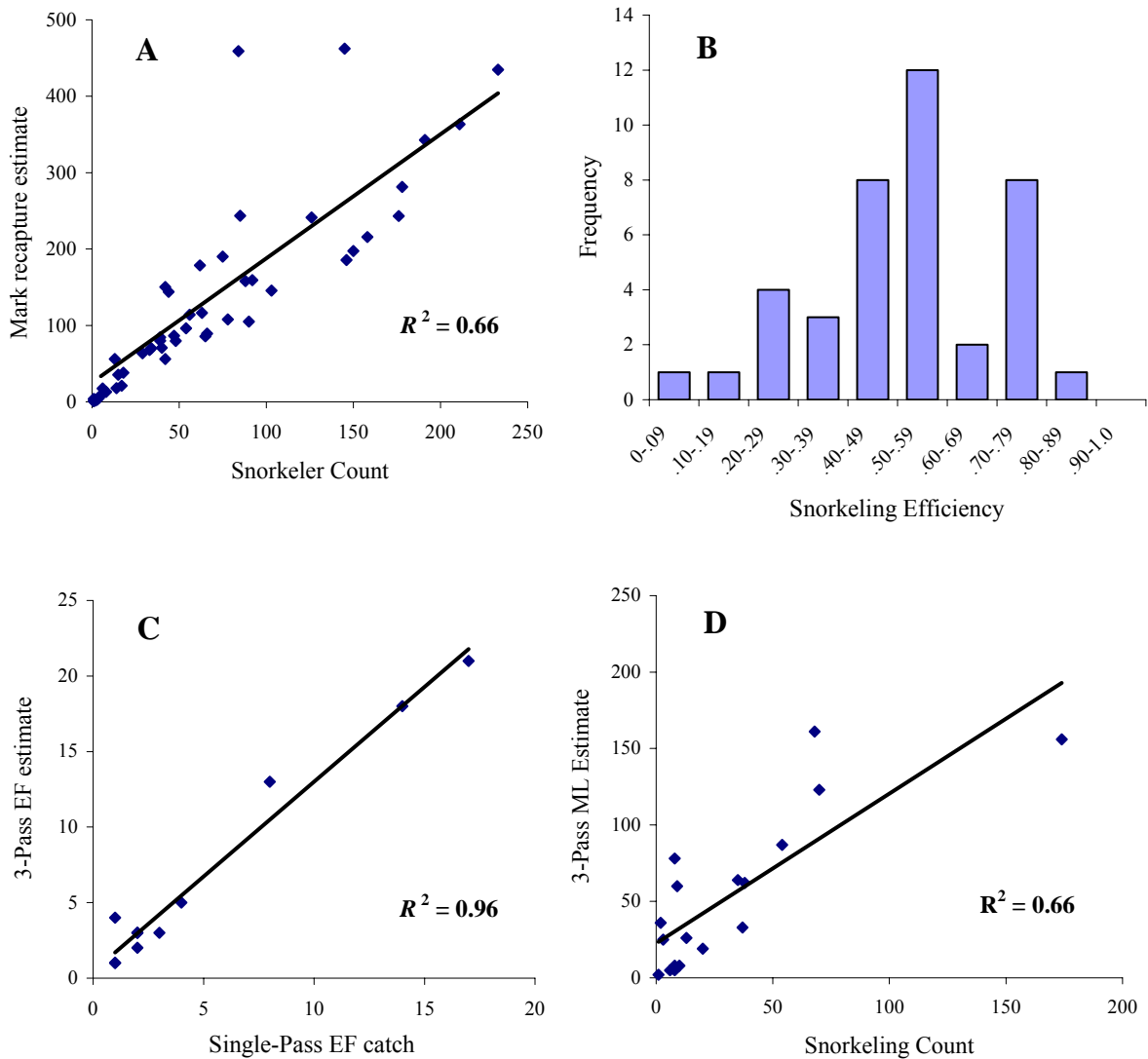


Figure 2. Graphs A and D are scatter-plots of snorkeling counts of Chinook fry versus Petersen mark-recapture estimates ($n = 40$) and three-pass electrofishing depletion estimates ($n = 18$), respectively, at snorkeling calibration sites in the lower Thompson River basin (not the same sites). Graph B is a histogram showing the distribution of snorkeling efficiency estimates (snorkeling counts divided by mark-recapture estimates) derived from the data shown in graph A. Graph C is a scatter-plot of single-pass electrofishing catches versus three-pass electrofishing depletion estimates ($n = 11$) at shallow riffle sites in smaller tributaries.

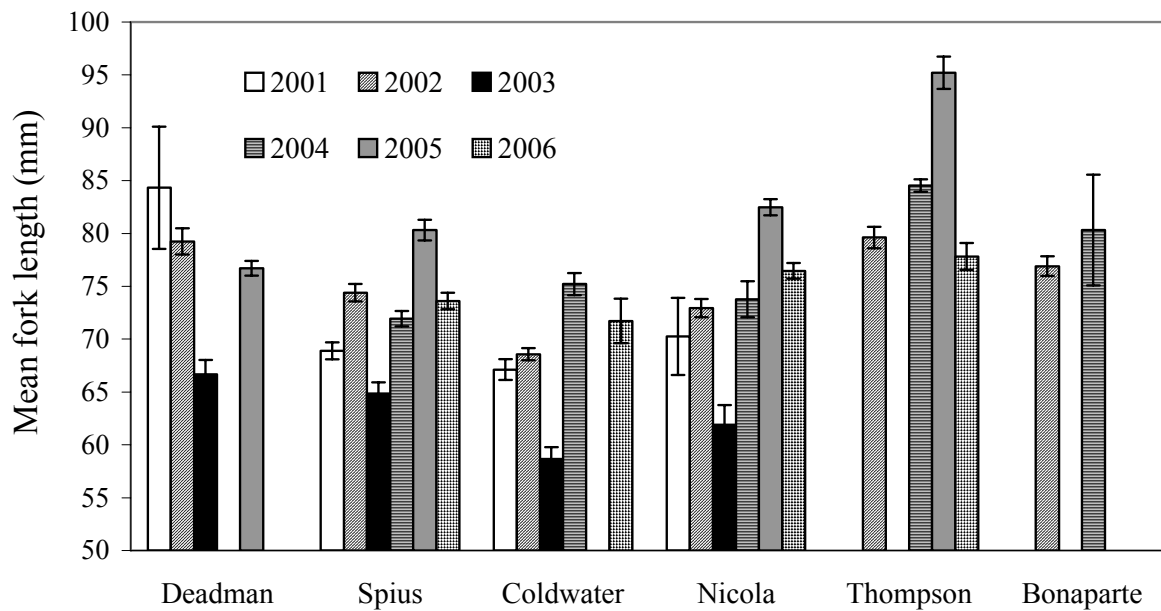


Figure 3. Mean annual fork lengths of Chinook fry in the six study streams in the lower Thompson River basin during October 2001-2006. Error bars represent ± 1 standard error of the mean.

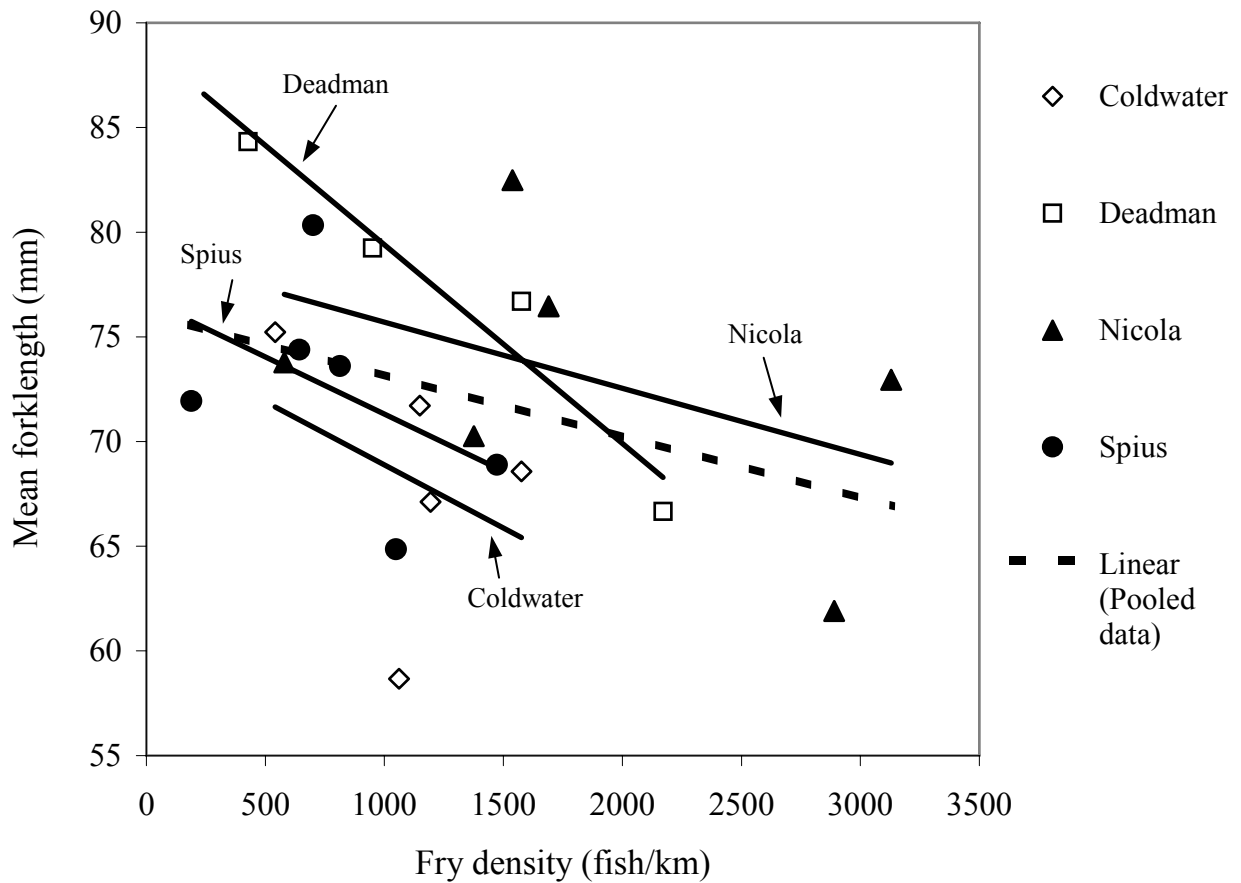


Figure 4. Scatter plots of mean annual fork length versus mean annual density for Chinook fry in four tributaries of the lower Thompson River. Solid lines show regression fits of mean length to fry density for individual streams and dashed line shows fit for pooled data from all streams.

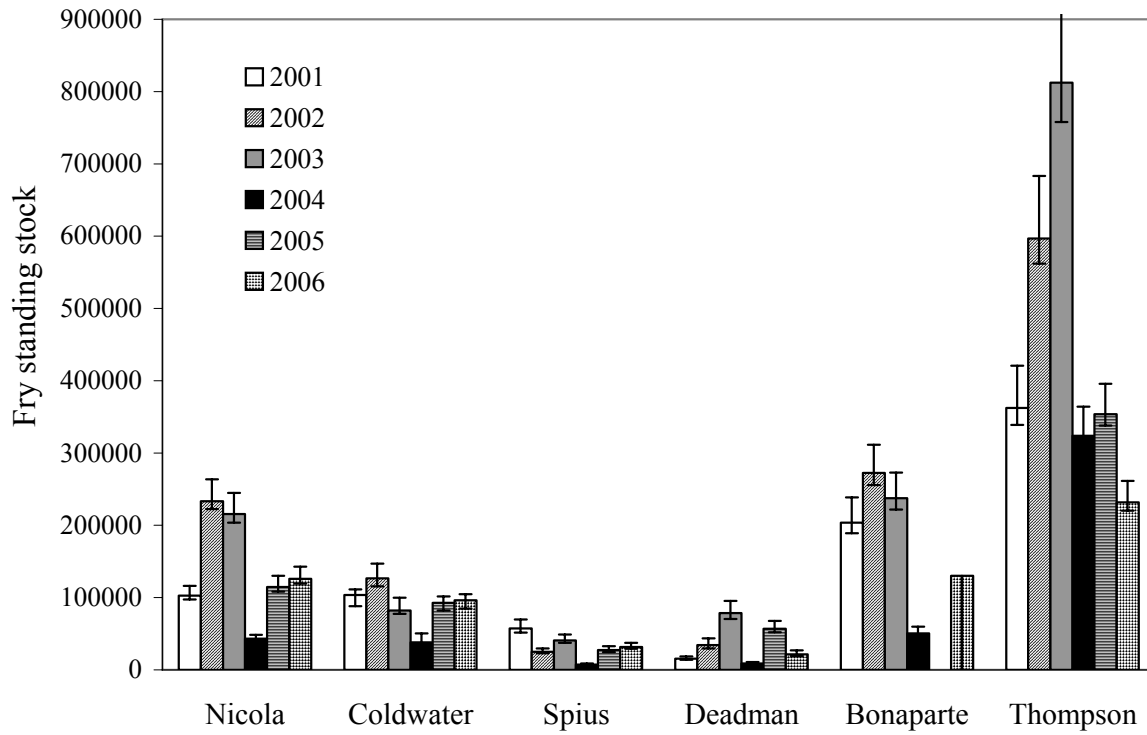


Figure 5. Chinook fry standing stocks by stream in the lower Thompson River during 2001-2006. Error bars represent 95% confidence intervals.

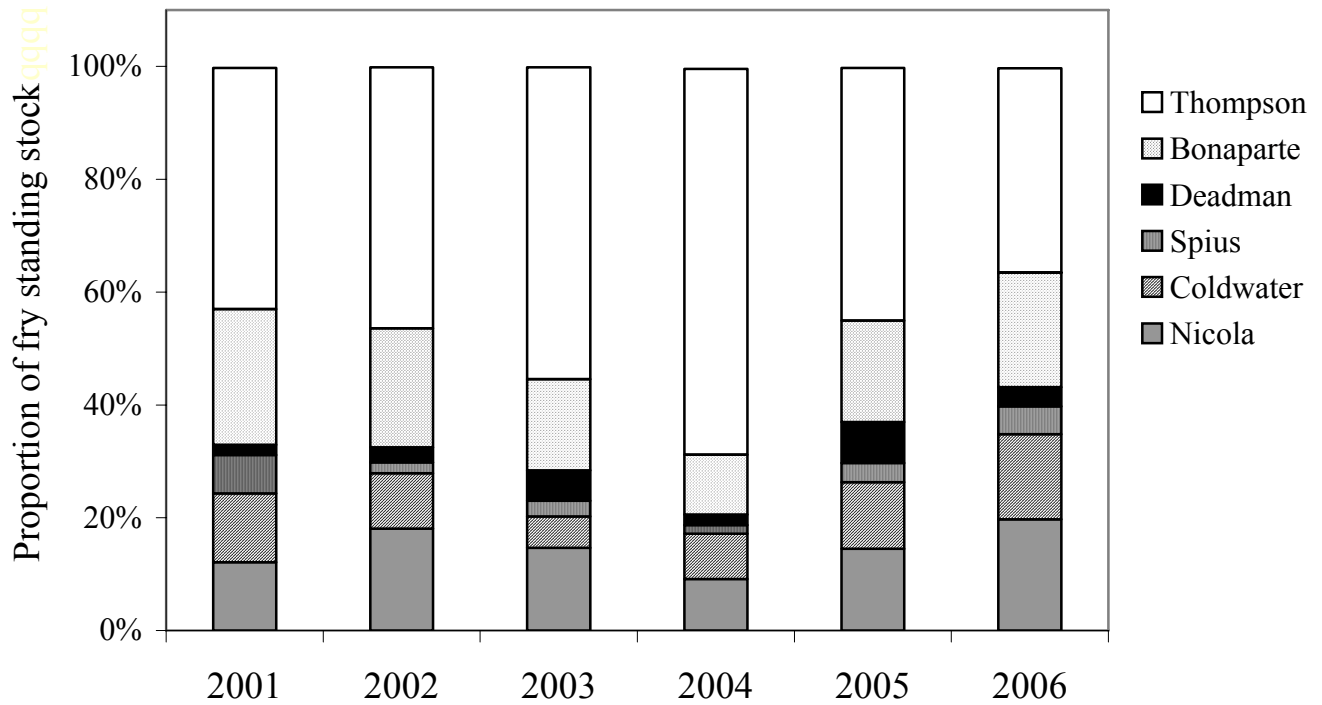


Figure 6. Relative contribution of each study stream to total Chinook fry standing stock in the lower Thompson River basin during 2001-2006. The value for the Bonaparte River in 2005 is simply the average for other years, as high streamflows precluding surveying Bonaparte River in that year.

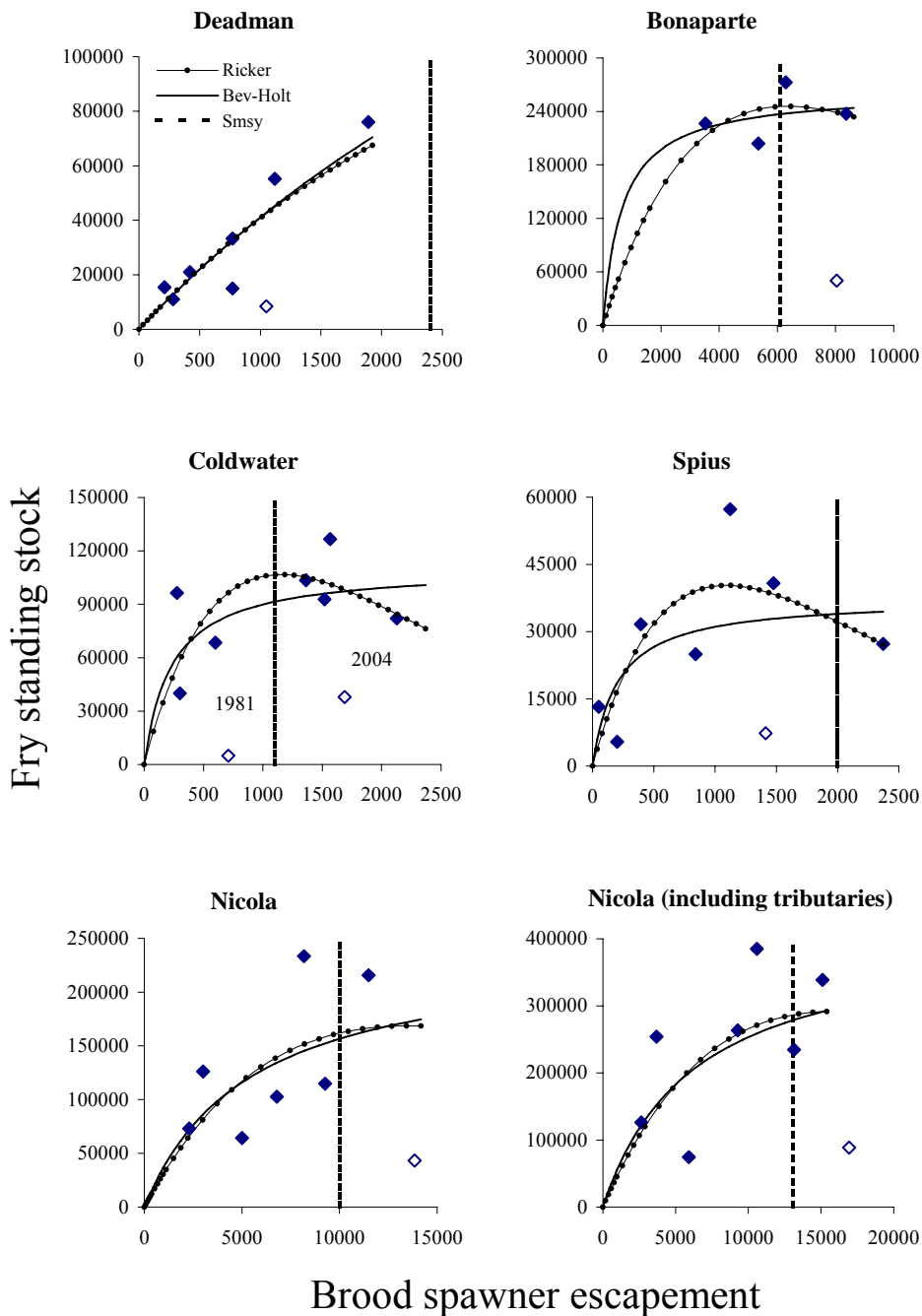


Figure 7. Ricker and Beverton-Holt stock-recruitment relationships for Chinook salmon in Thompson River tributaries. Vertical dotted line shows predicted S_{msy} from Parken et al. (2006) watershed area model. Open diamonds represent brood escapements and fry numbers for 2004; these data were considered outliers and excluded from the stock-recruitment analysis, as was data from the Coldwater River in 1981 (see Section 3.8).

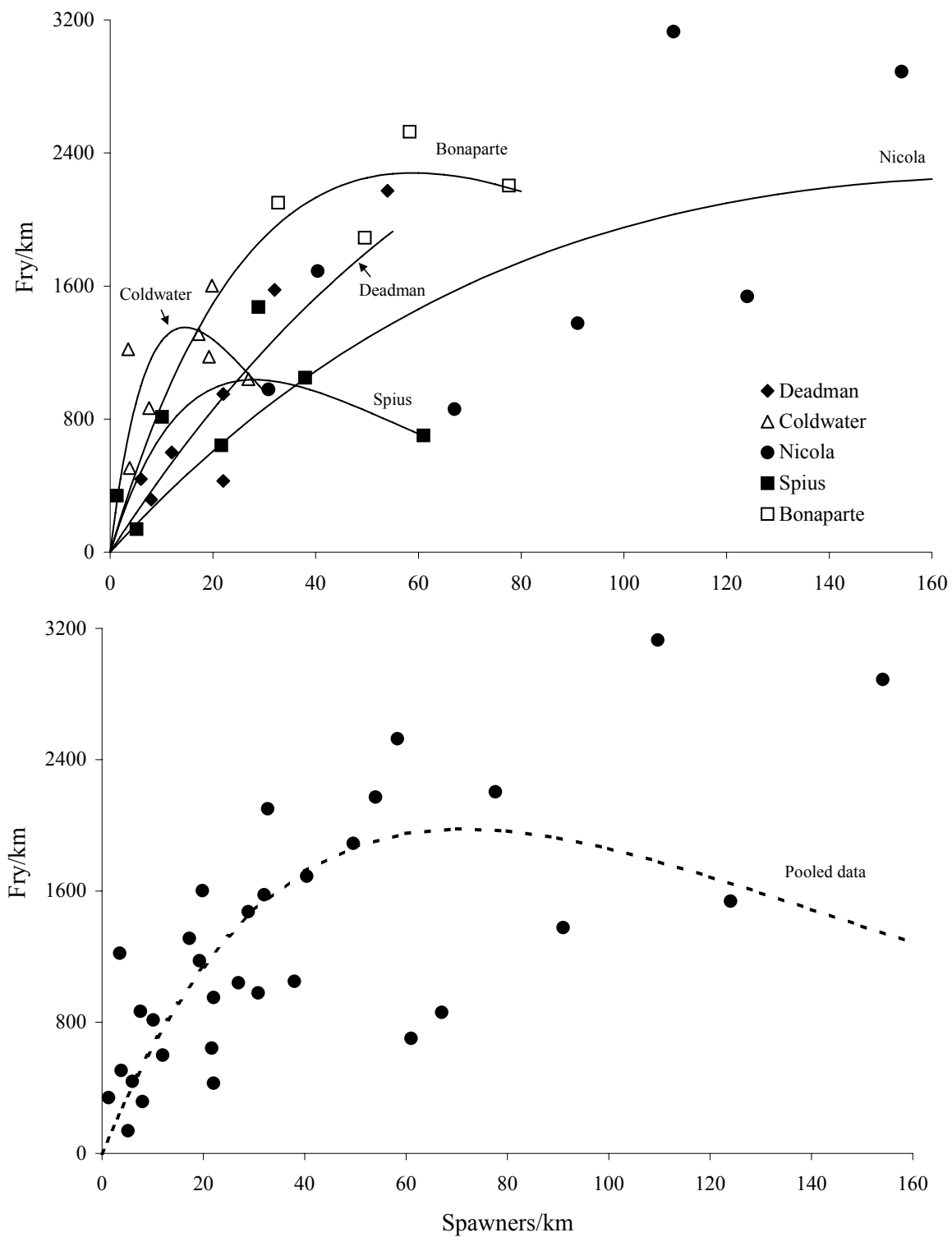


Figure 8. Ricker stock-recruitment relationships for Chinook salmon populations in individual Thompson River tributaries. Lower graph shows the predicted stock-recruitment relationship when data from the all tributaries are pooled. Data for 2004 were excluded, as were 1981 data for Coldwater River (see Section 3.8).

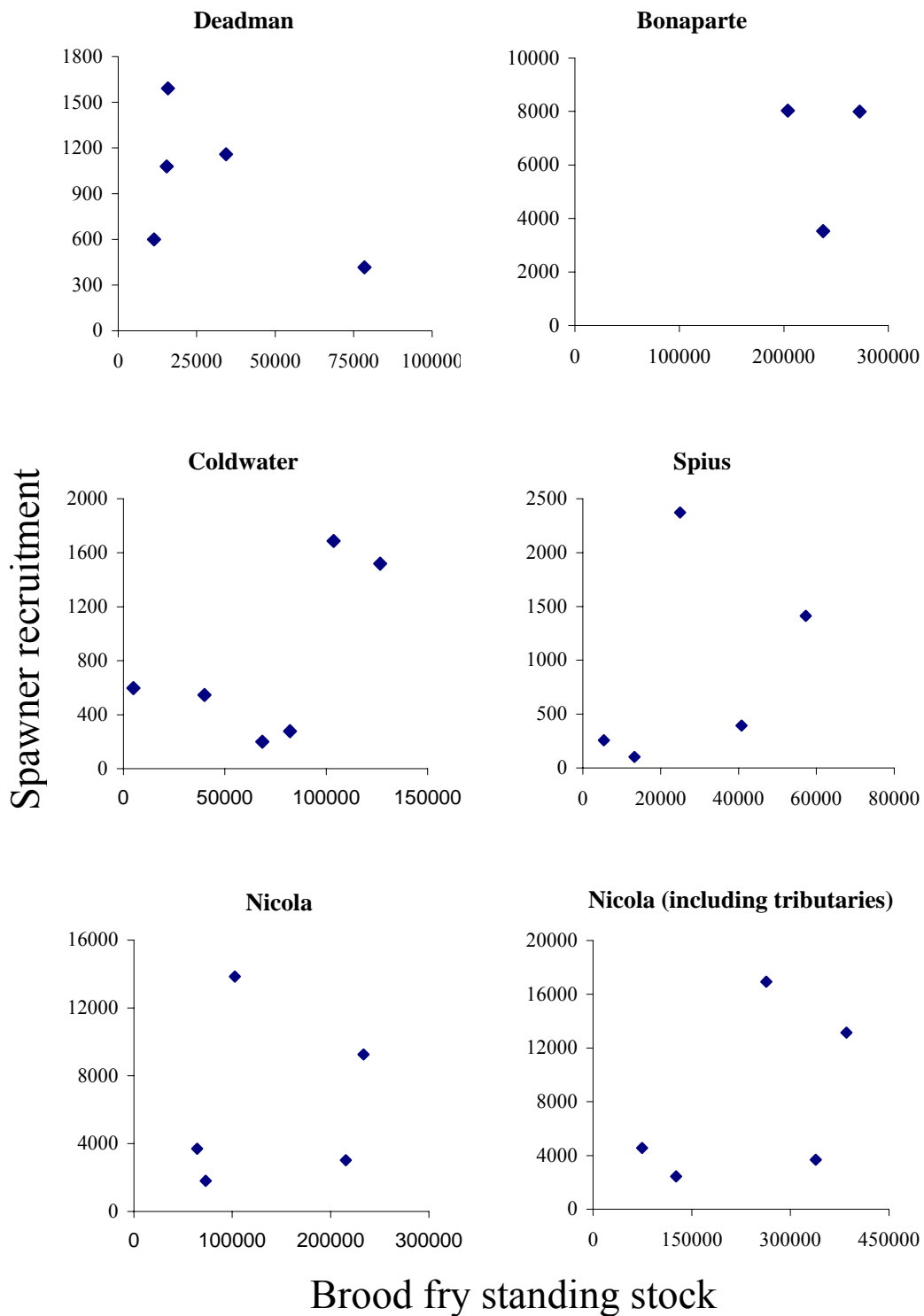


Figure 9. Chinook spawner recruitment resulting from brood fry standing stocks in individual Thompson River tributaries during 2001-2003 and various years prior to 1992. See Table 8 for data sources.

Appendix 1. Mark-recapture statistics for Chinook fry at 40 snorkeling calibration sites in six streams in the lower Thompson River during 2004-2006.

Site	Stream	Year	Temp.	Visi- bility	Total marks	Total recoveries in site	Total recoveries upstream	Total recoveries downstream	Total recoveries	Snork- eling count	Petersen estimate <i>N</i>	Snorkeling efficiency (count/ <i>N</i>)
b4*	Bonaparte	2004	5.5		2	1	0	0	1	5	9	0.56
b10_extended*	Bonaparte	2004	4.8		15	7	0	0	7	33	68	0.49
b15_extended	Bonaparte	2004	4.7		16	7	0	0	7	29	64	0.45
cw13*	Coldwater	2004	9.6		4	1	0	0	1	6	18	0.34
cw15a*	Coldwater	2004	9.3		18	10	0	0	10	40	71	0.56
cw17	Coldwater	2004	10.2		15	4	0	0	4	44	144	0.31
cw18a*	Coldwater	2004	11.4		11	4	1	0	5	18	38	0.47
cw19*	Coldwater	2004	10		3	0	0	0	0	13	56	0.23
cw4a	Coldwater	2004	10		13	7	0	0	7	54	96	0.56
cw8	Coldwater	2004	8		12	8	1	0	9	65	86	0.76
sp10	Spius	2004	9.5		8	4	0	0	4	47	86	0.54
sp2	Spius	2004	9.4		15	7	0	0	7	34	70	0.49
sp5	Spius	2004	9.4		13	5	0	1	6	39	80	0.49
sp9	Spius	2004	9.4		19	10	0	0	10	63	116	0.54
sp1A	Spius	2005	7.5	5	16	5	0	0	5	85	244	0.35
sp6	Spius	2005	7.5	5.5	15	10	0	1	11	66	89	0.74
spG	Spius	2005	5.5	8	12	7	0	0	7	48	80	0.60
cr15+ext	Criss	2005	8.1	3.5	19	7	1	0	8	15	36	0.42
cr5A	Criss	2005	6.6	3.5	14	11	1	0	12	90	105	0.86
d1	Deadman	2005	9.2	3	15	7	0	0	7	56	114	0.49
d10+ext	Deadman	2005	10.2	2	14	10	0	0	10	78	108	0.72
d13	Deadman	2005	9	6	12	9	0	0	9	42	56	0.75
d3	Deadman	2005	9.4	3.5	16	12	0	0	12	150	197	0.76
d6	Deadman	2005	9.4	3	21	13	0	0	13	178	281	0.63
n10A	Nicola	2005	9.1	2.5	18	9	0	0	9	126	241	0.52
n16	Nicola	2005	10	1.7	23	13	0	0	13	211	363	0.58
n22	Nicola	2005	10	2	24	12	1	0	13	191	343	0.56
n24	Nicola	2005	10	2.5	21	15	0	0	15	176	243	0.72
n10	Nicola	2006	8.5	4.5	26	4	0	0	4	84	459	0.18
n12a	Nicola	2006	8.5	4.5	25	12	1	0	13	233	435	0.54
t31	Thompson	2004	10		16	5	0	0	5	62	179	0.35
t33_extended	Thompson	2004	10.3		13	3	0	0	3	42	151	0.28
t39	Thompson	2004	10.3		23	12	0	1	13	92	159	0.58
t41	Thompson	2004	9.7		18	12	0	1	13	158	216	0.73
t42	Thompson	2004	9.7		13	9	0	0	9	103	146	0.71
t43_&_guard_rai	Thompson	2004	9.7		15	8	0	0	8	88	158	0.56
t45	Thompson	2004	10.3		18	7	0	1	8	39	84	0.46
t41	Thompson	2005	11	4.5	23	17	0	1	18	146	186	0.79
t42	Thompson	2005	11	4.5	24	8	1	0	9	75	190	0.39
tGR	Thompson	2005	11	4.5	18	5	0	0	5	145	462	0.31

Appendix 2. Summary of electrofishing and snorkeling count data collected at calibration sites in smaller tributaries of the lower Thompson River during 2001-2002. Three-pass maximum likelihood depletion estimates were used to provide estimates of single-pass electrofishing efficiency in shallow riffle strata and snorkeling efficiency in run strata.

Year	Site	Single-pass electrofishing catch	3-pass ML removal estimate	1-pass capture efficiency	Snorkeling count	Snorkeling efficiency (count/ML est.)
Calibration of snorkeling counts in runs						
2001	cr2	2	5	0.40	6	1.20
2001	cw13	46	62	0.74	38	0.61
2001	cw17	58	78	0.74	8	0.10
2001	cw2	23	33	0.70	37	1.12
2001	sp1	49	87	0.56	54	0.62
2001	sp6	100	161	0.62	68	0.42
2001	spD	6	8	0.75	8	1.00
2001	spE	20	26	0.77	13	0.50
2002	cr2	13	19	0.68	20	1.05
2002	cw18A	93	123	0.76	70	0.57
2002	cw22	21	25	0.84	3	0.12
2002	cw23	47	60	0.78	9	0.15
2002	cw5	8	8	1.00	10	1.25
2002	d12	2	2	1.00	1	0.50
2002	d17	7	36	0.19	2	0.06
2002	sp10	42	64	0.66	35	0.55
2002	sp9	126	156	0.81	174	1.12
2002	spJ	4	5	0.80	8	1.60
2001*	d10	1	1	1.00	11	11.00
2002*	spD	1	1	1.00	0	0.00
Calibration of single-pass electrofishing catches in riffles						
2001	cr1	2	3	0.67		
2001	cw14	2	3	0.67		
2003	cw25	1	4	0.25		
2001	cw3	1	1	1.00		
2003	cw3	4	5	0.80		
2003	cw26	17	21	0.81		
2001	sp4	8	13	0.62		
2003	sp4	2	2	1.00		
2005	spK	1	1	1.00		
2005	spM	3	3	1.00		
2005	cr1	14	18	0.78		

Appendix 3a. Estimated Chinook fry standing stocks, 95% confidence intervals, average percent relative errors (\pm CI), and densities (numbers/km) for 27 reach/habitat type strata in the lower Thompson River basin during 2001-2003.

Stream reach	Habitat type	2001					2002					2003				
		Standing stock	Lower CI	Upper CI	\pm CI	Fry /km	Standing stock	Lower CI	Upper CI	\pm CI	Fry /km	Standing stock	Lower CI	Upper CI	\pm CI	Fry /km
Lower Bonap.	rifle	18,096	15,834	22,422	18%	1,144	17,668	15,341	22,476	20%	1,117	49,950	40,523	68,392	28%	3,157
	run	164,857	149,451	197,870	15%	3,836	188,533	173,213	220,497	13%	4,387	119,910	109,693	141,892	13%	2,790
Upper Bonap.	rifle	8,336	6,566	12,015	33%	340	9,405	7,699	12,600	26%	384	1,567	1,187	2,316	36%	64
	run	12,521	10,762	15,678	20%	511	56,852	47,535	74,795	24%	2,321	66,114	55,747	85,600	23%	2,699
Lower Cold.	rifle	8,343	7,147	17,203	60%	536	17,372	12,128	29,862	51%	1,117	11,267	7,322	17,907	47%	724
	run	37,436	31,217	45,875	20%	3,034	65,396	54,489	77,668	18%	5,299	29,163	26,336	41,373	26%	2,363
Upper Cold.	rifle	9,897	6,450	14,932	43%	347	5,512	4,637	9,322	41%	163	5,007	4,842	10,206	54%	176
	run	47,889	36,080	46,916	11%	2,119	39,183	35,806	45,038	12%	1,734	36,710	32,715	42,221	13%	1,624
Deadman	rifle	1,144	923	1,695	34%	68	2,068	1,662	3,135	36%	123	3,399	2,721	5,231	37%	202
	run	14,319	12,864	17,305	16%	740	32,251	27,604	41,310	21%	1,668	75,051	67,007	91,869	17%	3,881
Lower Nicola	rifle	32,638	28,853	39,986	17%	1,173	95,039	86,432	112,052	13%	3,415	57,564	49,994	72,278	19%	2,068
	run	53,128	48,494	62,841	14%	2,333	109,612	100,891	128,495	13%	4,814	116,364	105,802	137,779	14%	5,110
Upper Nicola	rifle	4,659	3,625	7,442	41%	455	13,334	10,232	22,045	44%	1,301	8,724	6,683	14,142	43%	851
	run	12,229	10,665	15,154	18%	889	15,459	13,255	19,635	21%	1,124	32,901	28,913	41,033	18%	2,392
Lower Spius	rifle	3,131	2,541	4,732	35%	711	2,352	2,007	3,285	27%	534	2,815	2,280	4,294	36%	639
	run	13,194	11,457	16,576	19%	5,751	13,489	11,755	16,984	19%	5,879	15,792	14,104	19,187	16%	6,883
Upper Spius	rifle	3,003	2,195	5,128	49%	155	0	0	0	0%	0	4,862	3,812	7,595	39%	252
	run	37,974	32,457	49,047	22%	2,948	9,150	7,856	11,731	21%	710	17,339	14,302	23,288	26%	1,346
Thomp. (T1)	bar	13,976	12,109	17,417	19%	1,681	12,327	10,456	15,910	22%	1,483	12,220	10,321	15,885	23%	1,470
	dfs	11,012	9,544	13,794	19%	966	20,846	17,618	26,840	22%	1,828	17,426	14,434	23,036	25%	1,528
	embay	28,391	24,197	36,297	21%	1,432	48,040	39,943	63,122	24%	2,424	38,900	32,125	52,123	26%	1,963
Thomp. (T2)	bar	41,984	35,598	55,277	23%	2,924	80,602	69,541	102,023	20%	5,613	87,529	73,324	115,634	24%	6,095
	dfs	26,502	23,421	32,338	17%	1,330	53,066	46,266	66,444	19%	2,663	117,281	96,939	155,573	25%	5,887
	embay	47,078	41,085	58,418	18%	2,374	187,509	156,145	247,074	24%	9,457	312,863	260,805	407,392	23%	15,779
Thomp. (T3)	bar	8,964	7,754	11,487	21%	2,512	20,651	17,820	26,004	20%	5,788	30,993	26,702	39,134	20%	8,687
	dfs	39,987	33,595	52,339	23%	3,764	45,733	38,999	58,828	22%	4,305	62,176	52,400	80,711	23%	5,853
	embay	144,173	119,535	193,313	26%	12,121	127,931	108,773	165,126	22%	10,756	132,839	114,701	167,845	20%	11,168
Total or mean		835,609	815,535	918,804	6%	1,832	1,286,388	1,262,974	1,404,312	5%	2,820	1,468,472	1,424,648	1,616,134	7%	3,219

Appendix 3b. Estimated Chinook fry standing stocks, 95% confidence intervals, average percent relative errors (\pm CI), and densities (numbers/km) for 27 reach/habitat type strata in the lower Thompson River basin during 2004-2006.

Stream or reach	Habitat type	2004					2005					2006				
		Standing stock	Lower CI	Upper CI	\pm CI	Fry /km	Standing stock	Lower CI	Upper CI	\pm CI	Fry /km	Standing stock	Lower CI	Upper CI	\pm CI	Fry /km
Lower Bonap.	riffle	1,330	188	3,028	107%	84	ND ¹	-	-	-	-	21,507	9,430	38,177	67%	1,359
	run	22,496	13,400	37,051	53%	523	ND	-	-	-	-	170,410	110,129	253,137	42%	3,965
Upper Bonap.	riffle	0	0	0		0	ND	-	-	-	-	11,462	8,138	17,016	39%	468
	run	26,510	16,849	40,773	45%	1,082	ND	-	-	-	-	23,086	7,839	47,058	85%	942
Lower Cold.	riffle	2,593	0	3,723	72%	167	ND	-	-	-	-	5,899	3,304	11,213	67%	379
	run	19,809	14,938	28,605	34%	1,605	57,146	46,156	75,570	26%	4,631	45,411	31,743	66,225	38%	3,680
Upper Cold.	riffle	1,123	0	7,198	320%	39	ND	-	-	-	-	6,909	1,845	13,171	82%	242
	run	14,466	7,599	29,038	74%	640	19,797	13,493	27,283	35%	876	38,168	18,743	55,687	48%	1,689
Deadman	riffle	1,613	0	3,810	118%	96	3,325	1,019	6,972	90%	198	4,061	0	9,126	112%	242
	run	7,123	2,421	14,794	87%	368	53,619	26,280	98,695	68%	2,773	17,557	6,474	38,646	92%	908
Lower Nicola	riffle	10,381	5,668	18,676	63%	373	39,090	25,004	59,380	44%	1,405	50,345	26,605	85,968	59%	1,809
	run	28,287	22,238	37,956	28%	1,242	69,998	45,641	101,919	40%	3,074	57,539	34,130	91,604	50%	2,527
Upper Nicola	riffle	0	0	0		0	611	0	1,256	103%	60	3,312	0	8,265	125%	323
	run	4,590	2,878	6,873	44%	334	5,043	2,366	8,260	58%	367	14,838	9,434	23,203	46%	1,079
Lower Spius	riffle	1,415	509	3,027	89%	321	ND	-	-	-	-	6,058	4,523	9,557	42%	1,375
	run	4,511	3,026	6,495	38%	1,966	8,607	7,003	11,729	27%	3,751	11,928	7,939	16,661	37%	5,199
Upper Spius	riffle	0	0	0		0	1,404	0	3,618	129%	73	159	0	435	137%	8
	run	1,411	0	3,780	134%	110	15,199	4,889	30,885	86%	1,180	13,521	3,031	23,733	77%	1,050
Thomp. (T1)	bar	7,880	4,871	12,891	51%	948	15,428	4,608	29,984	82%	1,856	14,652	9,868	21,736	41%	1,763
	dfs	8,143	2,180	11,916	60%	714	11,389	8,382	15,904	33%	999	10,946	8,126	14,979	31%	960
	embay	45,352	8,842	86,347	85%	2,288	8,039	0	14,131	88%	406	21,893	3,197	52,654	113%	1,104
Thomp. (T2)	bar	44,144	35,624	57,813	25%	3,074	61,621	33,664	101,117	55%	4,291	44,481	31,964	67,720	40%	3,098
	dfs	45,896	19,148	86,422	73%	2,304	39,471	23,363	68,127	57%	1,981	32,234	12,831	65,907	82%	1,618
	embay	96,569	66,056	135,646	36%	4,870	81,361	55,496	109,756	33%	4,103	58,297	26,764	127,816	87%	2,940
Thomp. (T3)	bar	12,519	6,303	24,042	71%	3,509	18,516	10,764	29,276	50%	5,190	7,107	4,337	10,287	42%	1,992
	dfs	20,085	8,573	39,193	76%	1,891	31,096	11,854	55,160	70%	2,927	8,108	5,255	11,952	41%	763
	embay	42,975	36,179	55,537	23%	3,613	86,741	61,471	144,335	48%	7,293	33,879	14,547	55,631	61%	2,848
Total or mean		476,042	421,160	567,663	15%	1,044	770,850 ²	629,642	912,060	18%	1,690	728,093	646,495	881,957	16%	1,596

¹ ND indicates no data were collected due to high flows

² Chinook fry standing stocks were approximated for missing strata to compute 2005 total standing stock (see Section 3.6)